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EFFECT OF DIFFERENT MANAGEMENT INTENSITIES ON NATURAL AND HUMAN-INDUCED CARBON FLUXES IN A GOLF COURSE IN NORTHERN ITALY

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Acronyms

CO_2eq: Carbon dioxide Equivalent
CE: Carbon Equivalent
EC: Eddy Covariance
GPP: Gross Primary Production
GHGs: Greenhouse Gases
GWP: Global Warming Potential
IRGA: Infra Red Gas Analyser
LCA: Life Cycle Assessment
NECB: Net Ecosystem Carbon Balance
NEE: Net Ecosystem Exchange
NEP: Net Ecosystem Production
NPP: Net Primary Production
OC: Organic Carbon
OM: Organic Matter
PAR: Photosynthetically Active Radiation
P_{net}: Net Canopy Photosynthesis
P_g: Gross Photosynthesis
R_{eco}: Ecosystem Respiration
R_c: Canopy Respiration
SC: Small-chamber Enclosure

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Abstract

A thorough understanding of the terrestrial ecosystems' role on global carbon (C) cycle became crucial in last decades due to the climate change issues. Their response to global warming can result either in a positive or negative feedback on the radiative forcing caused by greenhouse gases (GHGs) emissions.

Environmental factors and managing policies have a significant effect on vegetation and soil properties, affecting soil organic carbon (SOC) pools in both extensive land uses (agricultural land, forests, semi-natural systems, etc.) and urban landscapes (urban forests, turfgrasses). Ecological approaches for the study of C fluxes have been widely adopted to study the C sequestration potential of natural unmanaged and managed ecosystems. Terms such as net ecosystem exchange (NEE) and net ecosystem carbon balance (NECB) have become critical index of the state and of the dynamics for different systems, including turfgrasses. These vegetated areas are ubiquitous in urban landscape and their role on C cycle is increasing important also due to the considerable footprint related to their management practices.

Several approaches have been proposed to assess C dynamics of urban turfgrass, changing on temporal and spatial scale. The present study critically review the available literature to compare the published data on C fluxes in turfgrasses taking in account the limits of each methodology and the meaning of the different estimates obtained. Few studies did use micro-meteorological methods such as eddy covariance to estimate NEE in turfgrasses, due to the usually small area and patchy distribution of these systems. Different approaches have been more widely used, each one with advantages and limits, and showing different estimates. The use of small surface chambers leads in most cases to positive NEE (CO_2 emission), while soil chrono-sequences and models almost always report negative NEE (CO_2 sequestration).

Carbon fluxes in turfgrasses resulted affected by natural and human-induced factors such as species composition, environmental conditions, site characteristics, former land use and agronomic management.

In order to study the effect of different management intensities on turfgrass, we estimated NEE in a hole of a golf course in the province of Verona (Italy) during one year (August 2012-September 2013) with a small chamber enclosure approach (SC). Whole-canopy gas exchange measurements were performed with a portable infrared gas analyser (IRGA) equipped with a closed transparent chamber. Surveys were conducted every two weeks in different playing areas, grouped in three management categories: high- (HI), medium- (MI) and low-intensity (LI) management. The SC approach likely resulted in an underestimation of the C uptake of the turfgrass because of the perturbation of the environmental conditions inside the chamber during the measurements.

All the maintenance categories showed similar trends but different daily CO_2 fluxes during the studied period likely due to different inputs and disturbances affecting biogeochemical cycles, combined also to the different leaf area index (LAI).

Net ecosystem exchange has shown to be related to the seasonality of turfgrass, following temperatures and physiological activity. Significant temporal variations were observed in CO_2 fluxes ranging from -2.46 to 4.78 µmol CO_2 m^{-2} s^{-1} (respectively C sink and source). Generally on the growing season CO_2 fluxes to the atmosphere exceeded C sequestered through photosynthesis. High respiration rate could be associated to the availability of subtle-broken organic matter (OM), water and warm temperatures. The CO_2 uptake was mainly controlled by PAR, while ecosystem respiration during the night was function of temperature.

The cumulative NEE for LI, MI and HI resulted 0.071, 0.057, 0.036 t C ha^{-1} yr^{-1} respectively, showing a significant decrease (therefore an increase in the annual C budget) with the increase of the intensity of management, and picturing a system near to a steady state for C dynamics.

In the final part an estimation of CO_2, CH_4 and N_2O direct emissions due to fossil fuel consumption for turfgrass upkeep was performed, based on machinery consumption and international guidelines for GHGs inventories. Using the above cited maintenance categories, turfgrass emissions due to maintenance operations resulted 0.846, 0.134 and 0.146 t CE ha^{-1} yr^{-1} respectively for HI, MI and LI. Even if maintenance activities increased C sequestration, they were at the same time responsible for C emissions.

Further studies need to be conducted to consider the C potential of trees and shrubs in order to complete the C budget of green urban areas and contrast GHGs emissions related to maintenance activities.

CHAPTER 1
Introduction

1.1 The Anthropocene

If an observer looked the Earth from a distance and followed its development for billions of years, he would find our planet weird and interesting. He would realise that in these years strange changes are happening and he would not understand the causes. He would note that temperature arises, atmosphere becomes more opaque, and a huge hole has been opening on the external ozone layer that envelops the globe. The observer is too far to understand details, but he deduces that on the shallow layer of the Earth a new power is altering appearance and equilibrium. He would have no doubt to be in front of the beginning of a new geologic epoch. If he knew that the main cause of these changes is the humanity, he would name the new epoch "Anthropocene" (Crutzen, 2005).

Crutzen (2005) places the beginning of this new era in 1784, with the invention of the steam engine. This discovery permitted an exponential exploitation of resources, generating the development of the industrial era and the resulting negative effect on environment. Other authors, more cautiously place the start of Anthropocene in the 1950's, with the sudden increase of scientific evidence regarding ecological problems connected to human activity (Steffen, et al., 2007). Other authors again claim that 10,000 years ago, with the development of agriculture, humans started to alter the atmosphere, as seen in the ice core analysis on Vostok Lake (Keeling and Whorf, 2005).

1.2 Climate change and the carbon cycle

Global climate change is nowadays unequivocal, and since the 1950s, many of the observed changes are unprecedented over decades to millennia (IPCC, 2013). The changes involve atmospheric gases concentration, atmospheric and surface temperatures (Figure 1.1) and, consequently, the diminution of the amounts of snow and ice coupled with the rise of sea level. Since 1990, scientific data and reports about climate change, produced worldwide, are periodically delivered by the Intergovernmental Panel on Climate Change (IPCC) into the Assessment

Reports. The last document was published in the end of 2013 and brought into focus the physical science basis to understand the current global situation.

The atmospheric concentrations of the most important greenhouse gases (GHGs) have increased to levels unprecedented in at least the last 800,000 years. In 2011 the concentration of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) were 391 ppm, 1803 ppm and 324 ppm, and exceeded the pre-industrial levels by about 40%, 150%, and 20%, respectively. Carbon dioxide concentration has increased primarily because of the fossil fuel burning emissions and secondarily because of the net land use changes emissions (IPCC, 2013).

Natural and anthropogenic processes that alter the Earth's energy budget, affecting the atmosphere composition, are the drivers of climate change. The strength of these drivers is quantified as radiative forcing. The largest contribution to total radiative forcing is related to the increase of the CO_2 atmospheric concentration provided by human activities. Of the cumulative anthropogenic CO_2 emissions from 1750 to 2011, 555 Gt C, 40% have accumulated in the atmosphere, 30% have been taken up by the ocean and 30% have accumulated in natural terrestrial ecosystems (IPCC, 2013).

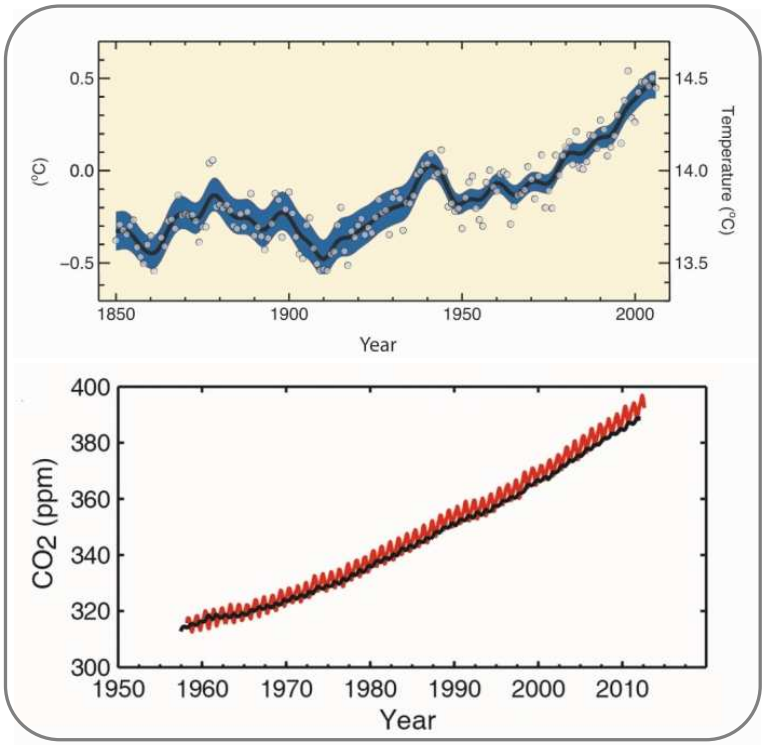


Figure 1.1: Observed changes in: atmospheric concentrations of CO_2 since 1958, above (IPCC, 2013) and global average surface temperature, below (IPCC, 2007).

In conclusion, data showed into the summary for policymakers of the fifth IPCC assessment report (IPCC, 2013) confirm that:

- it is extremely likely that human influence has been the dominant cause of the observed global warming since the mid-20th century;

- the evidences for human influence have grown since 2007;

- continued emissions of GHGs will cause further warming and changes in all components of the climate system;

- the projections of changes in the climate system show the alteration of C cycle and the further increase of CO_2 in the atmosphere;

- most aspects of climate change and of global warming consequences will persist for many centuries even if emissions of CO_2 were stopped.

1.3 The footprint of landscape components: managed turfgrass

The issue regarding the needs of future generations and the conservation of natural resources include also the preservation of landscape. Urban and suburban greening represents important elements for achieving global scale sustainable results. The "green system" participates to the macro-system, interacting with continuous matter and energy exchanges. The numerous benefits provided are well known and widely verified by several scientific researches (Laghai and Bahmanpour, 2012).

Even if green areas represent an important index for sustainable development of our cities, they can result not-sustainable. The absence of an adequate design and the intensive management can transform potential benefits (economical, social and environmental) into costs.

Despite a characteristic of natural systems is their capacity for self-maintenance, sustainable urban forests require human intervention. In fact anthropogenic activities transform site conditions (air pollution, over-building, etc.), compromising vegetation luxuriance. Furthermore, the quality of human interventions (for example appropriate tree care practices, right distances among planted trees, etc.) is not always appropriate, affecting the general sustainability of these areas.

In conclusion, urban greening does not always equal sustainable increase. Clark (1997) defines sustainable urban forests as "*the naturally occurred and planted trees in cities which are managed to provide the inhabitants with a continuing level of economic, social, environmental and ecological benefits today and into the future*". The first working model for urban forests (Clark and Matheny, 1998) provides the three fundamental components to reach the goals above mentioned: vegetation resource, community framework and resource management.

The definition of urban forest includes "*all woody and associated vegetation in and around dense human settlements*" (Miller, 1997), hence comprehending turfgrasses. Despite they are ubiquitous in urban landscape, their environmental benefits are not yet completely cleared. In addition to aesthetic value, they supply technical functions such as erosion control, absorption of pollutants, noise reduction and cooling effect on warm seasons.

Concerning global warming mitigation, several studies investigated the potential of turfgrass ecosystems to sequester C. As seen before, terrestrial ecosystems may play a fundamental role to mitigate climate change, accumulating CO_2 from the atmosphere. Several studies stated that turfgrass act as a C sink. However their role in continental and Mediterranean climates is not yet clearly understood, because environmental factors and management practices can strongly influence their overall C budget.

In Italy and Europe we observed in peer-reviewed literature the lack of studies that examined in depth the mechanisms underlying biogeochemical cycles and in particular C fluxes in turfgrass ecosystems. Different approaches and methodologies, above all originated from United States, have been proposed in the literature to assess turfgrass C sequestration potential. The range of experience and results is wide and there needs to summarize all the information provided from different studies. Among them just a small part provides estimates with direct measurement of C fluxes, and furthermore none shows annual overall C balances. Hence we pinpoint the necessity of a review of the state of the art, and of a field experiment that provide indication about the turfgrass behaviour within C cycle.

The importance of C sequestration and then of soil organic matter (SOM) is not limited to the global warming issue, but it is crucial also for turfgrass agronomic management.

Several greenkeepers, responsible for the maintenance of golf courses turfgrass, still prefer unnatural systems characterized by extremely sandy soils, low organic matter (OM) content and great quantities of external energy inputs (fertilizers, maintenance operations, agro-chemicals, irrigation, etc.). This approach preserves the amenity of turf, as it prevents water stagnation and fungal diseases, though it requires high environmental costs. Only in the past few years the conservation of OM has been boosted again in turfgrass soils. Difficulties for maintaining such unstable systems can be solved with a sustainable agronomic management that provide a more "living system", supporting the enhancement of SOM and of all that factors helping turfgrass to improve its resilience. The implementation of this approach could become important for improving C sequestration on urban landscapes.

1.4 Objectives

The primary aim of this thesis is thus to improve our understanding of the role of turfgrasses in the C cycle as effected by human and natural drivers and the interaction between them. The secondary aim is to achieve the best agronomical practices for a sustainable management, in order to respect the environment, satisfy human needs and limit the costs.

This is achieved by the following specific objectives:

1. Review the available literature on C cycling in turfgrasses, taking in account methodologies to measure C fluxes and stocks and analysing the factors driving their response to climate change.
2. Quantify the turfgrass C sequestration, through the estimate of the net ecosystem exchange (NEE) during one year, in different management intensities of a known and controlled system, such as golf course.
3. Estimate hidden anthropogenic C costs, related to the turfgrass maintenance.

1.5 Overview

Chapter 2 is a review of the existing literature concerning soil C sequestration in urban ecosystems, with particular attention to turfgrass. It presents an overview of the different methodological approaches for turf assessment as C sequestration driver. The analysis of the numerous studies provides important indications on the main variables involved in the natural and human-induced process that characterize these ecosystems.

Chapter 3 presents an experimental approach to assess net ecosystem exchange (NEE) in a golf course in northern Italy, as affected by environmental factors and different management intensities. A small-chamber enclosure approach was used to determine CO_2 exchange fluxes between atmosphere and turfgrass.

The Appendix analyses the hidden C costs due to the maintenance activities needed for maintaining turfgrass. At the end, general conclusions and recommendation are given in Chapter 4.

1.6 Bibliography

Clark J.R. and Matheny N.P. 1998. A model of urban forest sustainability: application to cities in the United States. Journal of Arboriculture 24: 112-120.

Clark J.R., Matheny N.P., Cross G. and Wake V. 1997. A model of urban forest sustainability. Journal of Arboriculture 23: 17-30.

Crutzen P. 2005. Benvenuti nell'Antropocene. L'uomo ha cambiato il clima, la Terra entra in una nuova era. Mondadori Saggi, strade blu, pp. 94.

IPCC. 2007. Summary for Policymakers. In: Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M.Tignor and H.L. Miller (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

IPCC. 2013. Summary for Policymakers. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S. K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Keeling C.D. and Whorf T.P. 2005. Atmospheric CO_2 records from sites in the SIO air sampling network. Trends: a compendium of data on global change 2009.

Laghai H.A. and Bahmanpour H. 2012. GIS Application in Urban Green space Per Capita Evaluation (Case study: City of Tehran). Annals of Biological Research 3: 8.

Miller R.W. 1997. Urban forestry: planning and managing urban greenspaces. New Jersey.

Steffen W., Crutzen P.J. and McNeill J.R. 2007. The Anthropocene: are humans now overwhelming the great forces of nature. Ambio: A Journal of the Human Environment 36: 614-621.

CHAPTER 2

Carbon sequestration of urban areas: different approaches and methodologies to estimate the carbon balance of turfgrass

2.1 Introduction

Carbon dioxide (CO_2) emissions from cities play a crucial role in the global carbon (C) cycle and in its feedback on climate change. Despite urban areas occupy less than 2.4% of global lands (Potere and Schneider, 2007), they are responsible for more than 80% of total CO_2 emissions. The majority of greenhouses gases (GHGs) emissions are due to energy consumption, material production and transportation, and land conversion from natural or semi-natural ecosystems into urban environment.

Despite human settlements have a footprint extending to continental scale, the majority of modelling studies focuses on the estimate of C fluxes through forests, grasslands and croplands, often omitting urban ecosystems (Pouyat, et al., 2003; Churkina, 2008). In developing countries, the high rate of human migration towards the cities and the resulting abrupt growth of urban areas, will increasingly affect the environment both at regional, considering the city as the centre of a huge important area outside its boundaries, and at global scale, concerning issues like climate change.

The continuous and increasing conversion of native ecosystems and agricultural soils to urban lands has been reported to have a negative effect on soil organic carbon (SOC) content (Pouyat, et al., 2006). Few examples showing the inverse process indicate that decades are needed to recovery C pools in a former agricultural land left to natural evolution, while recovery from urban land use may require more time or can never happen (Pouyat, et al., 2003). However, because their widespread diffusion and their importance in terms of extension, cities have been object of studies aimed to enhance their C storage potential in green belts and areas (Nowak, et al., 2002). In fact, despite their anthropogenic origins and variability, soils in urban areas still provide some important ecological services, maintaining the ability to store atmospheric C (Jo and McPherson, 1995; Pouyat, et al., 2006). It became thus crucial to understand the mechanisms underlying C fluxes in such complex systems and to estimate their potential for C sequestration. The available literature on the subject however, provides a number of different estimates of the C fluxes and/or C sink potential of these systems, based on

different approaches and methodologies. Each methodology gives results that consider fluxes originating from different processes. As a consequence, data are described by various terms (net ecosystem exchange, NEE, net ecosystem production, NEP, net ecosystem carbon balance, NECB, gross photosynthesis, P_g, net canopy photosynthesis, P_{net}, GHGs balance) which differ both in what they represent and in the units in which they are expressed.

The aim of this review is to present and summarize the current understanding of the role that turfgrass can have in the C cycle of golf courses and urban lawns as representative urban ecosystems. Different approaches and methodologies that have been proposed in the literature are critically analysed and their C sequestration rate estimates compared in order to extrapolate recommendations for the study and the management of such systems.

2.2 Carbon sequestration in urban ecosystems

2.2.1 Urban ecosystem towards sustaining self-regulating systems

The ecosystem theory classify urban areas as techno-ecosystems (Odum and Barrett, 2005), and ecologists have described the city as "*a heterotrophic ecosystem, highly dependent on large inputs of energy and materials and with a vast capacity to absorb emissions and waste*" (Alberti, 1996). Urban systems require large external areas to maintain the demands of energy and materials and to dispose of waste. Cities depend on countryside to provide almost all of its life-supporting resources, and conversely to pour into the same place its waste products. Material flows through urban areas are estimated to be so huge that areas 400-1000 times larger than the size of the cities are required (Decker, et al., 2003). Furthermore, in order to satisfy the huge demand of energy, urban industrial areas burn fossil fuels, produced in geological ages, with consequent large emissions of CO_2 and other GHGs. The amount of energy consumed per unit of area per year of an urban areas is 1000 or more times greater than that of a forest (Odum, 1997).

In conclusion the city can be compared to an organism with his own metabolism, continuously transforming and evolving. Urban sustainability is pursued guiding metabolic processes "*to facilitate the transformation of urban systems into sustaining self-regulating systems*"(Alberti, 1996).

2.2.2 Urban forestry

Since the beginning of the 1990s a number of studies, (e.g., Rowntree and Nowak, 1991; McPherson and Simpson, 1999; Nowak, et al., 2002; Escobedo, et al., 2011; Nowak et al. 2013), have demonstrated the importance of urban forestry for sequestering atmospheric C. Miller (1997) described urban forest as "*the sum of all*

woody and associated vegetation in and around dense human settlements, ranging from small communities in rural settings to metropolitan areas". Urban forestry is generally defined as "*the art, science and technology of managing trees and forest resources in and around urban community ecosystems for the physiological, sociological, economic, and aesthetic benefits trees provide to society*" (Konijnendijk, et al., 2006). Despite urban forests are less extended areas compared to natural and cultivated lands, they have an equal importance in contrasting global warming and should be therefore considered as a critical resource for policy makers, urban planners and managers (McPherson, et al., 1997).

Several cities of the United States have planned in last decades long-term urban-community forestry programs, in order to plant trees in large areas and educate citizens to good stewardship of urban greening (McPherson, et al., 1994; Nowak et al. 2008; Escobedo, et al., 2010). These experiences, combined to technical instruments (e.g. "Carbon Dioxide Reduction Through Urban Forestry: Guidelines for Professional and Volunteer Tree Planters")(McPherson and Simpson, 1999) permitted an exponential green expansion within the city and an valuable environmental effect. Together with the C storage, sociological, economic, psychological and aesthetic benefits are provided by green areas (Roy et al. 2012). Urban-community forestry programs, by involving and educating people to the urban greening, have obtained several positive indirect environmental results. Citizen involved in programs felt the role to control green spaces dynamics and furthermore applied the same easy rules for their private gardens (plant the right tree in the right place, act with adequate cultural practices, how to prune trees, etc). The citizen understanding of good agronomical practices has been important to improve the quality of urban green and then its environmental benefits (Colangelo, et al., 2006). In fact urban forests are not C sequestration systems a priori: different drivers can push the system to be either a C sink or a C source. Trees sequester and store C in their tissue at differing rates and amounts, depending on several factors, such as size at maturity, life span, and growth rate. At the same time, tree-care practices release C to the atmosphere by fossil-fuel combustion during maintenance, while dead trees eventually decompose releasing further CO_2. Thus, some of the C gains from tree growth can be offset by C losses to the atmosphere via natural process or produced from maintenance activities (Nowak, et al., 2002).

2.2.3 Turfgrass land use as carbon sequestration driver for urban areas

During the past decades, urbanization in west countries has proceeded at unprecedented rates and extent, converting large areas of vegetation into urbanized landscapes (Milesi, et al., 2005; Golubiewski, 2006). Nevertheless, this process have allowed the establishment of urban green areas, including turfgrass surfaces too.

Turf can be defined as "*a surface layer of earth containing a dense growth of grass and its matted roots*" (The American Heritage Dictionary, 2013). From an ecosystemic point of view, turfgrass includes the plant community, the soil rizosphere and the above ground organic matter (Turgeon, 1985). In a productive view, turfgrass is defined as "*a particular agricultural crop where the yield is not the removal part but what remains on the field*" (Cereti, 2002).

The more appropriate term for stating these surfaces in technical and scientific discussion is turfgrass, or at least turf, whilst non-specialised names such as grass and lawn are widely used in informal speech. A lawn is an area of land planted with grasses which are maintained at a short height and used for aesthetic and recreational purposes. Grasses are herbaceous plants with narrow leaves growing from the base. They include the "true grasses" of the Poaceae family, that constitute lawns and grasslands, as well as other species. These terms specify the function and the outwards characteristics.

Turfgrasses provide both suitable surfaces for sports and free time, and valuable green areas for monumental, architectonic and residential settlements. They also play technical functions such as erosion control, absorption of pollutants and particles, noise reduction, green cover for infrastructures (airports, roadsides, etc.) and cooling effect.

Turfgrasses are extensively used in Europe and United States urban landscapes, including residential garden, parks, commercial, institutional lawns, recreational facilities, green belts, sports fields, and golf courses. In the continental United States is estimated that potentially 16,381,200 ha of land area are cultivated with some form of lawn, which is three times larger than that of any irrigated cropland (Milesi, et al., 2005). This estimate, provided by the analysis of high resolutions aerial photographs, accounts for approximately 1.9% of the total US area. Another research concerning the US territory confirm that 16-20 million ha of urbanized land, or up to 18% of the land area in some regions, are covered with turfgrass (NASS, 2004).

If a number of studies about turfgrass total area have been carried out in the USA in the last decades (Vinlove and Torla, 1994), no specific data and researches are available for the European continent. However, several statistic data about urban green areas are available for most European countries.

Green space coverage of cities has been shown to vary markedly, averaging 18.6 per cent and ranging from 1.9% (Reggio Calabria, Italy) to 46% (Ferrol, Spain) (Fuller and Gaston, 2009). Per capita green space provision varied from 3-4 m^2 per person (Reggio Calabria) to more than 300 m^2 in Liége (Belgium) forming a clear spatial pattern at continental scale, with lowest provision in the south and east of Europe, increasing to the north and north-west (Fuller and Gaston, 2009). Whereas, considering the Italian cities, an average of 31.4 m^2 of urban green area per person was accounted in 2012 (ISTAT, 2012). Despite the lack of studies regarding turf extension in Europe, considering the presence of urban turfs proportional to green space areas, we assume that turfgrasses are ubiquitous in European cities.

Soil C dynamics in turfgrass have been examined in last decade in urban areas, such as golf courses (Qian and Follett, 2002; Bandaranayake, et al., 2003; Huh, et al., 2008; Qian, et al., 2010; Bartlett and James, 2011; Selhorst and Lal, 2011), residential lawns (Pouyat, et al., 2002; Golubiewski, 2006; Pouyat, et al., 2009; Raciti, et al., 2011), athletic fields (Townsend-Small and Czimczik, 2010) and urban lawns (Kaye, et al., 2005; Milesi, et al., 2005; Allaire, et al., 2008; Livesley, et al., 2010; Zhou, et al., 2012). Results from these studies indicated a strong capacity for these vegetative covers, used in different context and with different function, to sequester C in soil.

The relative high extension, their ubiquity and the high productivity rate make turfgrasses a potential tool for C sequestration in urban areas, although indirect costs related to the management play a crucial role in order to make them a sustainable or not-sustainable element of the new urban landscape.

In addition to ecosystem sustainability, understanding soil organic matter (SOM) accumulation is of great importance because have profound influence on soil fertility and soil structure. Soil organic matter content is of particular interest to turf managers because:

- changes in SOM influence the air-filled porosity, water retention, and percolation in the sand-based root zone, which is used intensively in putting greens, tees, and sports fields;

- SOM serves as a major reserve of plant nutrients, especially N, P, S, and K.

For these reasons, greenkeepers investigate SOM through soil analysis and are really careful in maintaining the right equilibrium between soil components and elements. On the other hand, limited information are available concerning the long-term soil organic carbon (SOC) changes in turfgrass systems.

2.2.4 Soil organic carbon dynamics in urban ecosystems

The change in SOC occurring during the urbanization of a site can be fundamental to understand the ecosystem response to urban land-use change. In urbanized landscapes, anthropogenic drivers dominate over natural drivers in the control of SOC dynamics (Kaye, et al., 2006).

The World Reference Base (WRB) classification for soils recognizes urban and industrial soils as technosols (Rossiter, 2007), "joining" the more wide definition of techno-ecosystems given by Odum (1997) to such systems. As a consequence of human disturbances, technosols are characterized by strongly altered profiles, and high heterogeneity (Hollis, 1991; Bullock and Gregory, 2009). This type of soils is the result of several variables affecting soil properties after conversion to urban land use of natural ecosystems. Among these factors, those that influence SOC can be divided in direct and indirect.

Direct factors include physical disturbance, coverage of soil through overbuilding and all the practices that alter the chemical and biological characteristic of soil, including management inputs (irrigation and fertilization).

Indirect effects of urbanization concern changes of environment process (biotic and abiotic), influencing soil development at long temporal scale (hundreds of years) (Pouyat, et al., 2003). Indirect effects include the increase of temperatures within cities, called "urban heat island", soil hydrophobicity, introductions of exotic species, atmospheric deposition of pollutants and stress effects of the urban environment on soil decomposers and primary producers.

All the above mentioned factors result in the so called "parcelization process", which is the conversion of vegetated ecosystems to urban land by modifying the system in an array of soil patches that range from natural to disturbed to landfill profiles, to covered soils. This process causes the alteration of management regimes that will affect the territory over the time. As a result of the parcelization, a wide spatial variability in the soils can be found within the urban environment, depending on previously and current land use. Some studies conducted in parks of big urban areas (Tokyo, New York, etc.), comparing soil analysis of random samples, confirmed a high variability also in small spaces (Cramer, 1993; Jim, 1998). Furthermore Pouyat and colleagues (1997) suggest that the presence and distribution of unmodified soil patches increase from highly dense populated areas to rural and natural areas. The same gradient is effective for population density, percentage of impervious surface, automotive traffic volume, etc. (Pouyat, et al., 2003).

Despite anthropic activity in urban areas can negatively affect SOC pools, its role in C cycle is primary and promoting soil C sequestration is an effective strategy to reduce atmospheric CO_2 (Lal, et al., 1999).

Carbon sequestration in soils can occur directly or through plant mediation. Direct soil C sequestration is the conversion of CO_2 into inorganic C compounds (carbonates), whereas, plant-mediated C sequestration implies CO_2 fixation through photosynthesis, followed by C transfer to the soil through root exudation and dead biomass litter (above and below ground), leading to SOM transformation and decomposition by soil biota.

Carbon budgets in vegetated urban soil are in a dynamic imbalance, determined by natural fluxes (photosynthesis, autotrophic and heterotrophic respiration, leaching, etc.) and anthropogenic fluxes (combustion of fossil fuel for maintenance activities, export of biomass off the system, etc.). In general, accumulation of SOC into ecosystems requires a positive imbalance between inputs and outputs of SOM stocks. When a positive disequilibrium is sustained over some period of time, then C sequestration occurs, with the system eventually achieving a new, higher steady-state. Schulze and colleagues (2000) suggest that even mature, late successional stage ecosystems, such as old growth forests, can continue to act as net sinks for atmospheric CO_2 until high ages.

In an ecosystem, several factors, such as climate, soil parent material and biota, time and human footprint drive the processes connected to vegetation and decomposition, and thus SOC formation and stabilization (Figure 2.1).

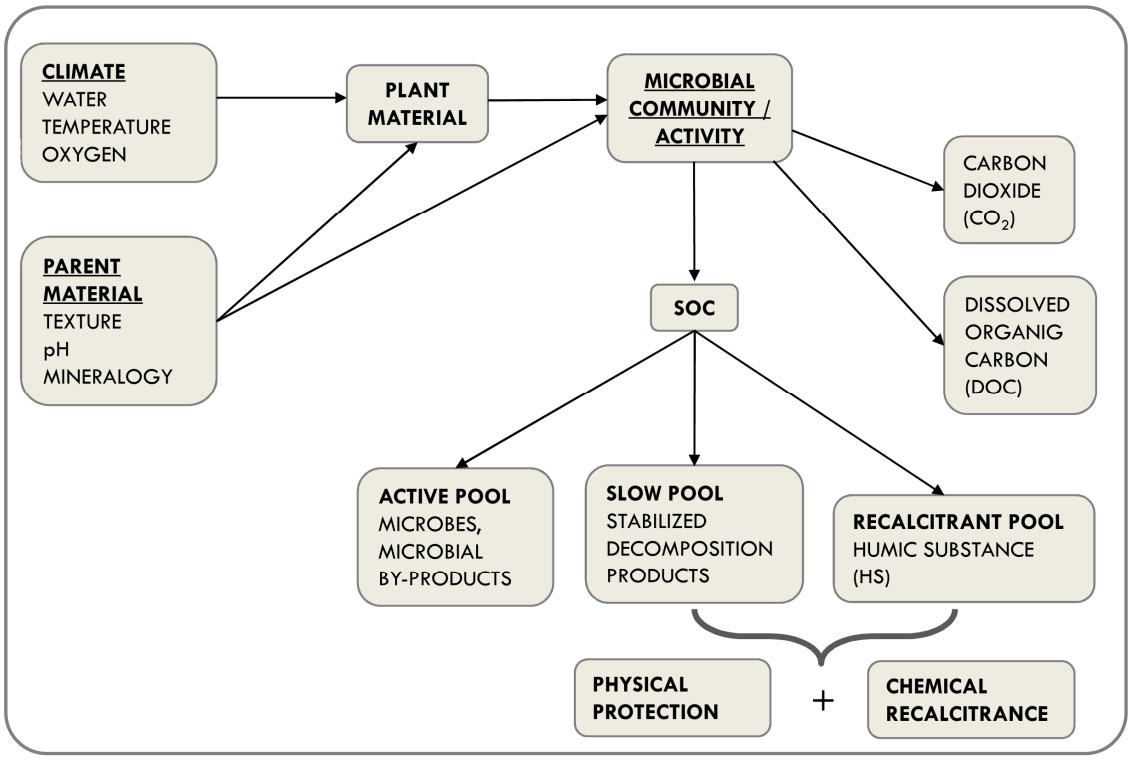


Figure 2.1: Factors affecting soil organic carbon (SOC) formation and stabilization. Reworked graphic version from Singh (2007).

The SOC constitutes three pools:

- one active pool which has the least residence time and consists of live soil microbes and microbial products short turnover time;

- one slow pool which has an intermediate residence time and represents stabilized decomposition products (characterized by physical protection and chemical recalcitrance against further microbial action);

- one passive pool which has a residence time of years (hundred and thousand) and is composed by stable humic substances (recalcitrant pools).

In urban soils, the strategies to increase soil C sequestration should work at two different levels: 1) increasing plant C inputs to the soil and 2) increasing C residence time in soils by enhancing its accumulation in the slow and passive pools.

1) Unlike semi-natural ecosystems, vegetation in the cities is always maintained by humans. Despite the negative effect related to anthropogenic emissions, cultural operations aimed to enhance the plant efficiency and the C fixation, such as N fertilization and irrigation, can increase plant productivity and accelerate biogeochemical cycles, resulting in a greater accumulation of SOM (Qian, et al., 2003; Milesi, et al., 2005). The productivity of managed urban vegetation could also increase because of changes in environmental drivers such as atmospheric temperatures and CO_2 concentration. The effect of all these factors on SOC however, will greatly depend also on climate and soil properties before urbanisation.

2) Two major mechanisms, bio-chemical alteration (humification) and physico-chemical protection, stabilise SOM and thereby control its turnover. The humification process decrease the bioavailability of C and thereby the microbial activity, increasing the stability of C within the system. The physicochemical protection of C in the micro-pores of soil aggregates is provided by the interaction of the soil mineral matrix and bio-molecules. If the soil mineral matrix is absent, such in the turfgrass thatch (shallow biological active horizon), the residence time entirely depend on the degree of recalcitrance defined by the chemical structure of SOC (Krull, et al., 2003).

The operative practices to enhance the C sequestration potential of soils include increasing the period under which the soil is vegetated, maintaining high standards of “photosynthetic activity” and minimising cultivation and other disturbances (Jastrow, et al., 2007).

2.3 Carbon sequestration in turfgrass: definitions, methodologies and estimates of the net ecosystem carbon balance (NECB).

2.3.1 Definitions

Schulze et al. (2005) summarize the original definition of an ecosystem given by Tansley (1935) by defining ecosystems as "*networks of interrelations between organisms and their environment in a defined space*". This definition does not describe boundaries, compartments or characteristics of the systems. According to Aber and Melillo (2001), terrestrial ecosystem studies regard the metabolism and physiology of unites of landscape, defining the movement of energy and material (water, nutrients, chemicals, pollutants). Every ecosystem can be conceptualized as a volume with defined top, bottom and sides. In terrestrial ecosystems the top of this defined volume is typically above the canopy and the bottom is below the rooting zone (Randerson, et al., 2002).

These concepts well fit the definition of turfgrass: in fact several processes (supported by the interaction of herbaceous species, microorganism and anthropogenic inputs) occur in a delimited space, concerning less than 50 cm, characterized by environmental variables. Golf courses turfgrasses represent controlled systems, with pools and fluxes relatively easy to conceptualize. Homogeneity of substrate, establishment and cultural operations, floristic composition and inputs amounts (irrigation and fertilization), result in an anthropic ecosystem with spatial uniformity of vegetational cover and roots system (Figure 2.2). Rhizosphere's volume and biomass are usually constant overtime, depending on turf species (different roots depth), turnover rate, microorganism activity and aeration frequency. Above ground biomass depends on canopy photosynthetic rate, and is influenced by season, climate, and maintenance activities, however it is usually maintained stable through mowing operations.

Figure 2.2: section of turfgrass showing spatial homogeneity of above- and below-ground biomass.

Photosynthesis is the process by which C and solar energy enter ecosystems: the total amount of C fixed in this process by autotrophic organisms is termed Gross Primary Production (GPP) and it is generally expressed per unit area and time. Plants reduce CO_2 with energy derived from the sun and produce organic compounds that contain C and binding energy. The fixed C follows different paths to be either quickly respired by plants metabolisms (autotrophic respiration) or transformed in organic matter (OM) which also follows different fates through the ecosystem and returns in circle with different timing. Carbon and energy enter, move through and leave the ecosystems, in associated forms. The biochemistry of photosynthesis governs the environmental controls over C inputs to ecosystem (Chapin III and Matson, 2011).

Despite the energy content and C concentration in OM is extremely variable at molecular scale, they are relatively constant for species and location at plant and organ scale. The energy content is about 20 KJ g^{-1} of ash-free dry mass, while the C concentration is about 45% of dry weight in herbaceous and 50% in woody tissues (Chapin III and Matson, 2011). Because of the relative constancy, C, energy and biomass have been used as units to evaluate ecosystems dynamics, varying among different studies.

Carbon uptake by photosynthesis (GPP), and respiration (autotrophic and heterotrophic) are usually the largest C fluxes between a terrestrial ecosystem and the atmosphere, their balance is termed net ecosystem production (NEP).

Net ecosystem production explicitly links the physiology of organisms to the C balance of the ecosystem where the organisms live. It is virtually impossible to measure NEP directly. However, gaseous exchange involved in the balance between GPP and respiration represent most of the C mobilized. The net CO_2 exchange between the atmosphere and the ecosystem, termed net ecosystem exchange (NEE), is therefore a reasonable approximation of NEP, when measured over short time periods (Chapin III, et al. 2006). NEE (negative) and NEP (positive) correspond to a C input to/from an ecosystem in a determined time. Another term used for the net rate of C accumulation is the net ecosystem carbon balance (NECB) which represents the whole ecosystem C balance from physical, biological and anthropogenic sinks and source (Chapin III, et al., 2006). It considers C input and output from the ecosystem over a specified time interval.

$$NECB=\Delta C/\Delta t \qquad \textbf{(eq. 2.1)}$$

Several fluxes can contribute to NECB:

$$NECB=-NEE+ F_{CO}+ F_{CH4}+ F_{VOC}+ F_{DIC}+ F_{DOC}+ F_{PC} \quad \textbf{(eq. 2.2)}$$

Where F_{CO} is net carbon monoxide absorption, F_{CH4} is net methane consumption, F_{VOC} is net volatile organic compound absorption, F_{DIC} and F_{DOC} are net dissolved inorganic and organic carbon input and F_{PC} is lateral transfer of OM (Table 2.1).

Table 2.1: Components of net ecosystem carbon balance (NECB) equation, elaborated from Chapin III, et al. (2006).

Abbrev.	Term	Value positive/ negative
NECB	Net Ecosystem Carbon Balance	sink/source
NEE	Net Ecosystem Exchange	source/sink
FCO	Net C monoxide absorption	absorption/efflux
FVOC	Net volatile organic C absorption	absorption/efflux
FDIC	Net dissolved inorganic C input	absorption/efflux
FDOC	Net dissolved organic C input	absorption/efflux
FCH4	Net methane consumption	absorption/efflux
FPC	Lateral transfer of OM (soil and leaf clippings removal, soil ameliorant apport)	input/output

Net ecosystem exchange is defined by micro-meteorologists as an exchange of C between the atmosphere and the ecosystem resulting in negative values when the system sequester C (photosynthesis prevails) and positive values when the system release C (respiration prevails). On the contrary, NEP and NECB are defined by ecologists as a measure of ecosystem productivity having opposite sign than NEE.

Net ecosystem carbon balance diverges from NEP when inorganic and organic C enters or leaves ecosystems in forms other than CO_2 or dissolved inorganic carbon (DIC). Important fluxes regarding turfgrass ecosystems include dissolved inorganic (DIC) and organic (DOC) carbon leaching, emissions of methane (CH_4), carbon monoxide (CO) and volatile organic carbon (VOC), and C entering and leaving turfgrass through maintenance activities. In particular, these last fluxes consider SOM removed through aeration, above ground biomass C removed through mowing (leaf clippings) and OM supplied with soil ameliorant. All the terms present in the equation change with temporal and spatial scale. Net biome productivity (NBP) (Schulze, et al., 2000) is NECB estimated at large temporal and spatial scales, otherwise is the temporal average of NECB over a heterogeneous landscape (Chapin III, et al., 2006).

2.3.2 Golf courses and turfgrass characterization

For better understanding the studies concerning soil C sequestration in turfgrass, it is necessary a brief introduction regarding the various situations and surfaces we can run into and their characteristic. If the terms “home lawn” and “urban turf" indicate common vegetation covers, with variable maintenance degrees and heterogeneous soil horizons changing from site to site, details regarding sport fields, and particularly golf courses, are not as much known.

The presence of different specific playing areas join all the golf courses, however every single layout worldwide make each particular golf course unique. In accordance with our aim of study, the main important sections in a golf course are the following: green, tee, fairway, rough, semi-rough and collar.

Green, or putting green, is the most important surface and generally is the evaluating criterion to judge the golf course maintenance degree. Greens are the areas where turfgrass has the most stressful conditions, because all the players step on these parts of the golf course. The high pressure that concentrates in such a small areas influences soil porosity and turf growth. The United States Golf Association (USGA) guidelines for green construction considers the use of sands and gravels in soil layers, in combination with a regular drainage system, in order to create a structure that supports intensive playing and at the same time allows air and water circulation for the correct development of the grass roots (Figure 2.3).

Figure 2.3: The construction of United States Golf Association (USGA) greens: soil profiles. Reworked graphic version from Eriksson (2012).

Tees are the starting platform of each hole. They are smaller areas than greens and, as the previous, they are exposed to higher pressure and people traffic. Greens and tees are strongly fertilized, mowed at low height and very frequently, approximately 200 and 100 cuts per year respectively (personal data related to cool-season grass). These surfaces require the preservation of a delicate balance, particularly for avoiding water-logging, the most important issue for turf cultivation in our climates. More over the adoption of other special cultural operations such as coring (removal of small cores of soil from the top soil layers) and topdressing (filling cores with sand) is necessary to maintain the turf to its optimal physiological conditions.

Different heights of grass are maintained between the teeing ground and green to increase playing difficulty (R&A, 2014). Fairway is the area between the tee and green where the grass is cut frequently (100 cut per year). The play in these areas is less intense.

"*The turfed area surrounding the putting green, tee and fairway is commonly referred as rough. The rough and non-playable areas represent more than 70% of the total golf course area wherein a diverse ecosystem rich in native naturalized plant and animal species are maximized*" (Beard, 2002). Usually the rough is mowed infrequently (40 cut per year) and not fertilized.

Semi-rough represents a transient area between fairway and rough and has intermediate characteristics. The collar is a ring of grass around the putting green. The height of the grass in the collar is typically around the mid-point between the height of the green and the height of the fairway grasses.

The turfgrasses can include two different typologies of species, mainly from the family of Poaceae, also called Gramineae or true grasses: cool-season and warm-season species, respectively with C3 and C4 carbon fixation metabolism.

The two groups have different optimal ranges of temperatures for physiological activities: for below-ground activity the optimal range is between 10 and 18°C for cool-season grasses and between 24 and 32 °C for the warm-season ones, while for the photosynthesis activity the ranges are between 15 and 24 °C, and between 30 and 37° C respectively (Croce, et al., 2006).

Cool season grass species are all available as seed, whereas warm season species are primarily available sod or stolons. The most common cool-season turfgrass species are *Lolium perenne*, *Poa pratensis*, *Poa annua*, *Festuca arundinacea*, *Agrostis stolonifera,* while the most common warm-season turfgrass species are *Cynodon dactylon*, *Paspalum vaginatum* and *Zoysia spp..*

Cool-season grasses are widely spread both in cool-humid climates and in cold semi-arid areas, while warm-season grasses originate from warm climates, both humid and arid. However, at certain latitudes, called transition zones, both categories can grow. In the transition zones warm-season turfgrass species suffer

from winter dormancy and loss of green colour, while for cool-season species is difficult to maintain growth rates during the warm summer. Therefore, transition zones areas in which to maintain quality turf is very challenging. Italy's climate fits almost entirely within the transition zone (De Luca, et al., 2008). Research carried out in Italy have shown the good adaptability of warm-season grasses around the N 43º parallel (Miele, et al., 2000) and also further north, around the N 45º parallel (Croce, et al., 2003; De Luca, et al., 2008). In the Italian climate, characterized by hot and dry summers, turfgrass establishment is increasingly carried out with warm season species as opposed to the more widespread cool season species (Croce, et al., 2003).

2.3.3 Methodologies

Different approaches have been proposed to assess the C dynamics of urban turfgrass. Canopy-chamber methods (small-chamber enclosures, SC) require to place the sample in a closed transparent chamber (Figure 2.4). The variation of gas concentration calculated over a specific period of time by an infrared gas analyser (IRGA) connected to the chamber, corresponds to the rate of gas exchange by the system (Hunt, 2003). A limitation of this approach is the alteration of the CO_2 concentration gradients between soil, vegetation and air, caused by the measure itself. Several publication (Bremer and Ham, 2005; Kutzbach, et al., 2007; Murphy, 2007) report systems, methodologies and mathematical models to avoid and limiting instrumental errors (see Chapter 3 for a more detailed description of chamber based estimation of NEE).

Eddy covariance methods (EC) calculate vertical turbulent fluxes within atmospheric boundary layers, measuring the exchanges of CO_2, water vapour, and energy between terrestrial ecosystems and the atmosphere (Figure 2.5). An EC sampling tower analyzes high-frequency wind, scalar atmospheric data series and yields values of fluxes, through an ultrasonic anemometer and IRGA.

Eddy covariance and SC are the most used methods to measure gas exchange between turfgrass and atmosphere. Direct measurements of land-atmosphere fluxes using the EC method is however challenging in turfgrass and lawn ecosystems due to the high heterogeneity of urban landscapes and the usually small dimensions of lawns. Even if measurements are conducted above an extended surface of turfgrass, the potential fluxes may be influenced by nearby CO_2 sources such as vehicular road traffic (Hiller, et al., 2011) or other vegetation typologies (Lewis, 2010).

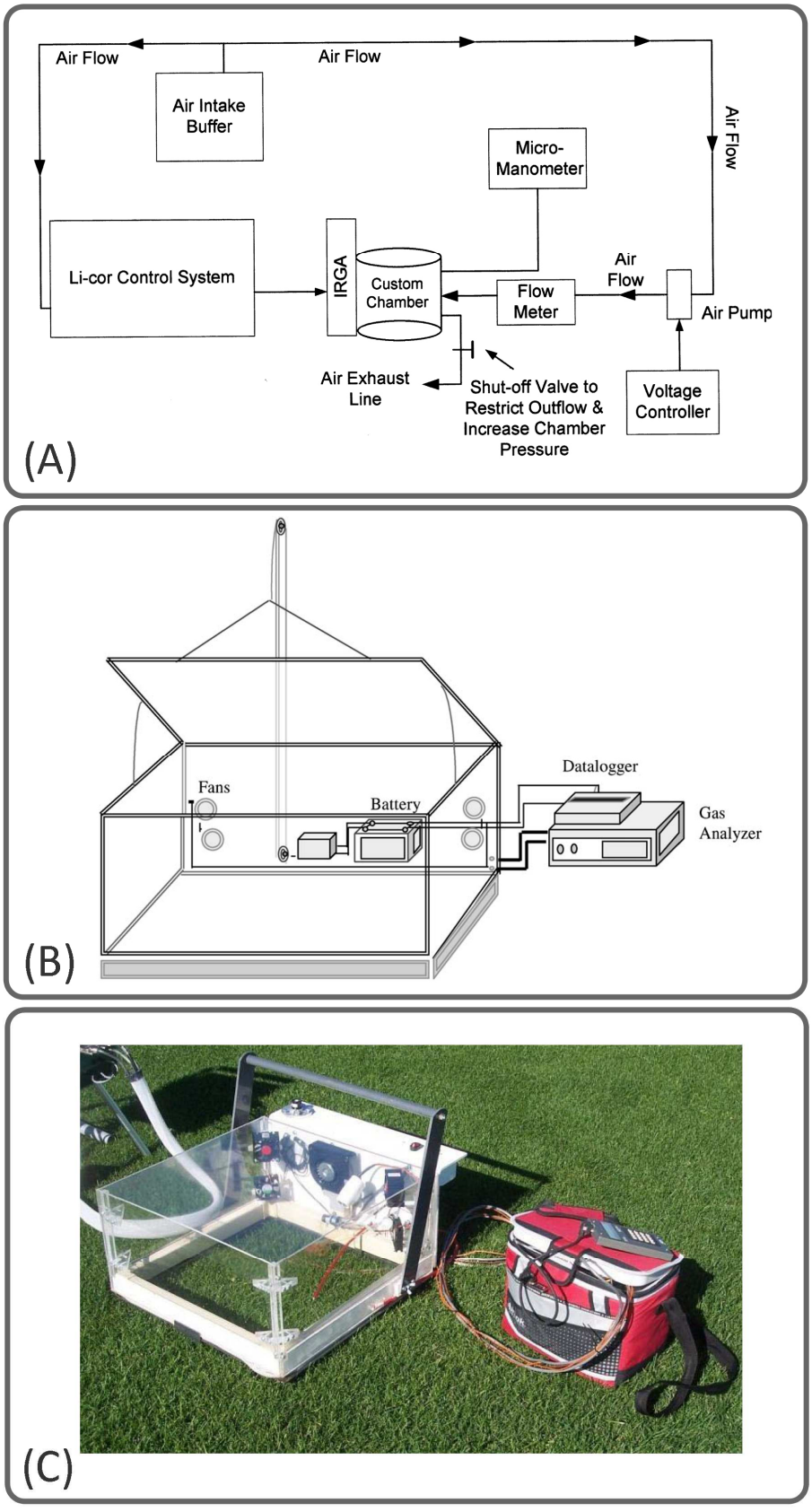


Figure 2.4: Small-chamber enclosure approach: (A) conceptual diagram illustrating the design of a pressurized chamber system (Bremer and Ham, 2005); (B) schematic representation of the automated closed-system (Steduto, et al., 2002); (C) picture of a custom gas exchange chamber (Lewis, 2010).

Several studies used the chrono-sequence approach for the estimate of soil C sequestration of turfgrasses. Changes in SOC occur slowly and annual changes are generally small (Qian and Follett, 2002). Since it would need decades of measurements to asses SOC trends, the chrono-sequence approach is often adopted, which consists in sampling soil in areas with turfgrass of different ages, assuming that all the systems analysed had the same dynamics. Using the space for time substitution is equivalent to study soil C stocks and dynamics over the lifespan of an ecosystem.

Data from chrono-sequences (SOC accumulation) are comparable with NEE because turfgrass C pools (biomass, thatch, SOC) are considered at steady state (Figure 2.6). The epigeum layer of turfgrass is regularly mowed, with recycling of leaf clippings, hence above ground biomass results constant. The thatch, that is the layer of dead turf material, and the below ground biomass are themselves in equilibrium with SOC, thanks to regular decomposition and roots-turnover. In accordance to these postulates, in a mature turfgrass the constant biomass and the lack of C inputs and outputs (in a simple representation of the ecosystem that cannot consider secondary fluxes as DOC, DIC, etc.) has as a consequence that SOC is the only pool that increases or decreases depending on turfgrass GPP. Thus the annual change of SOC is equal to NECB, or -NEE.

An approach not frequently used to assess SOC dynamics in turfgrass systems is the isotopic approach, based on the natural abundance of C isotopes such as ^{13}C. The photosynthetic pathways of C3 plants, such as those that constitute cool-season turfgrasses, discriminate against ^{13}C (1.1% of the C in the biosphere) differently from C4 plants, which prevail in warm-season turfs, thus resulting in different C isotope ratios ($^{13}C/^{12}C$) of the OM.

The mean $\delta^{13}C$ of warm-season and cool-season plant tissues are near -13‰ and -27‰, respectively. This difference in the natural abundance of ^{13}C has been often used to trace OC in the soil. For example, when cool-season turfgrass is established on fields that previously hosted warm-season plants, isotope technique can be used to determine the contribution of turfgrass to SOC accumulation.

Some ecosystem models (CENTURY, DAYCENT, BIOME-BGC) have been applied to turfgrass in order to simulate bio-geo-chemical processes (Figure 2.7). Furthermore, a model (CRANTURFC) based upon a Life Cycle Analysis (LCA) have been developed to understand the GHGs emission of turfgrasses and to consider the hidden C costs. Also the use of remote sensing combined to biophysical measurements of light use efficiency has been used to estimate the NPP of turfgrasses. Whatever the boundaries of the system, modelling needs physiological and technical data sets (respectively for natural and anthropogenic processes).

Changes in SOC can be compared with -NEE and NECB, in order to assess the role of turfgrass on global C cycles (C-sink or -source). The NPP estimates can be related to turf yield, but not to the C balance of the whole ecosystem as heterotrophic respiration has to be considered. In Table 2.2 all the values of NEE, NEP, NECB and SOC accumulation available in the published literature for turfgrasses have been summarized, grouped for methodology of estimation.

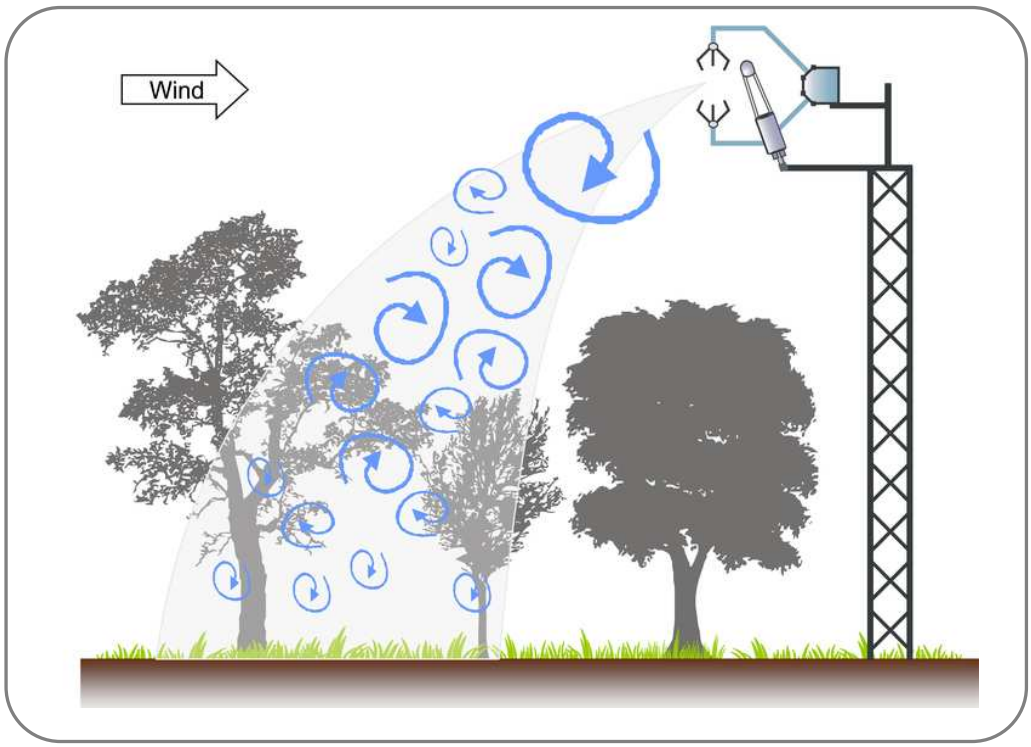


Figure 2.5: Eddy covariance approach to the measurement of CO_2 balance over unclosed ecosystem (Wolf, 2010).

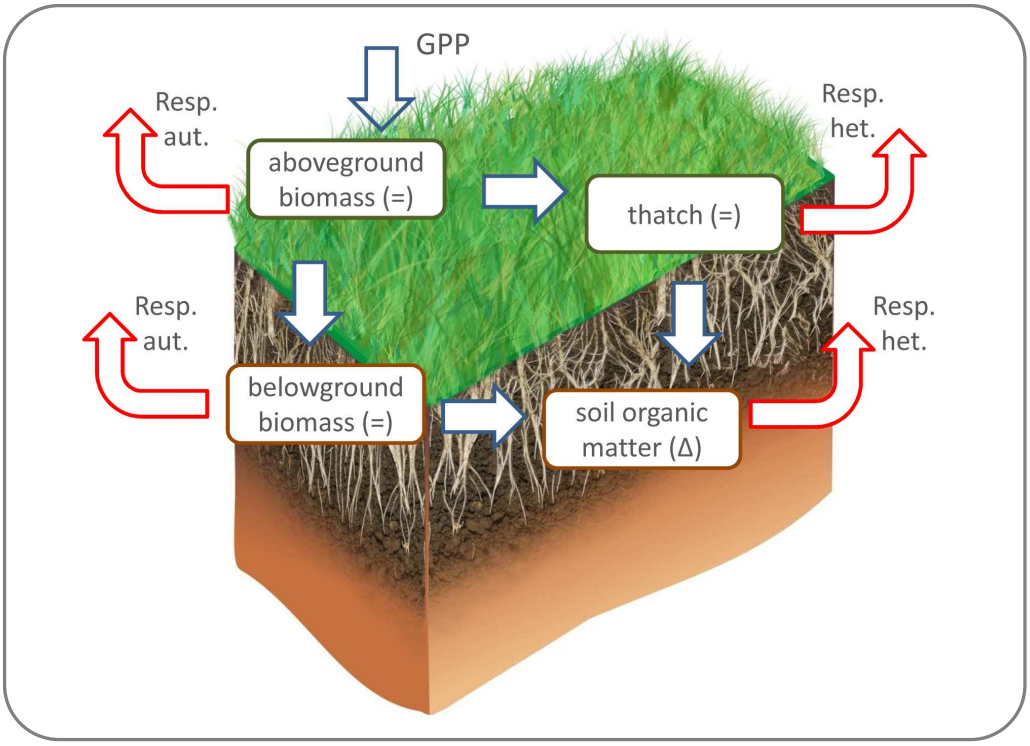


Figure 2.6: Representation of C pools (boxes) and fluxes (arrows) on turfgrass ecosystem. Symbol in brackets indicate changes (Δ) or steady state (=) .

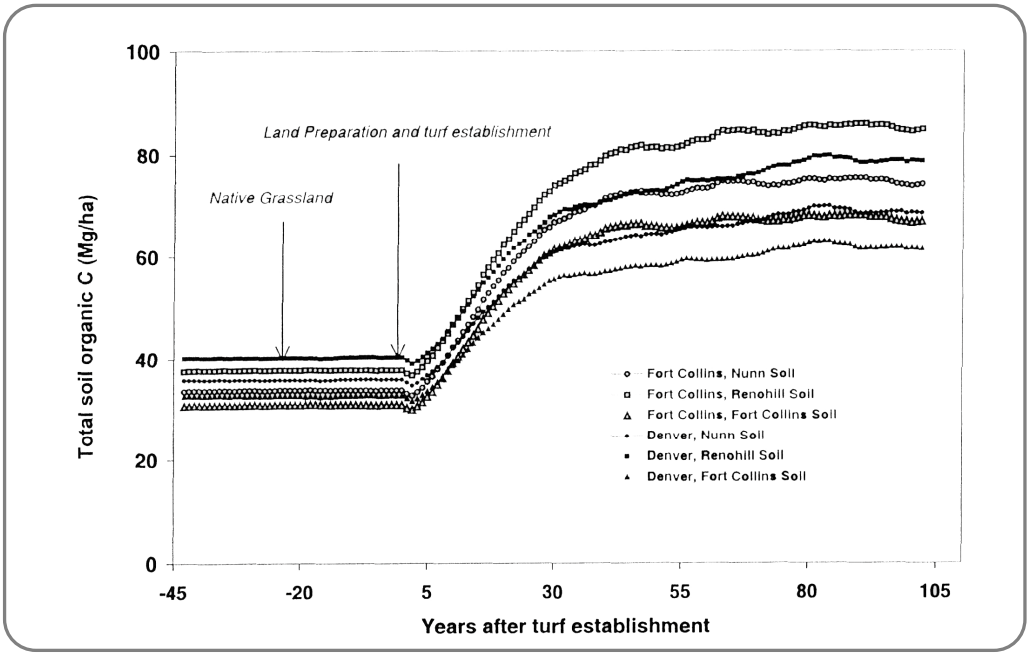


Figure 2.7: Representation of soil organic carbon (SOC) trends simulated by CENTURY model (Bandaranayake, et al., 2003).

Table 2.2: Turfgrass C sequestration estimates grouped by methodology of estimation as reported in the literature. ΔSOC= increase in soil organic carbon; NECB= net ecosystems carbon balance; NEE=net ecosystems exchange; NPP=net primary production. See section 2.3.1. for definitions and comparability of the values. All the values are expressed in t C ha^{-1} yr^{-1}. Site, previous land use, typology, species and depth investigated (where available) are reported. Letters in brackets near C sequestration estimates refers to different methods: BREB method (a); EC method (b); small-chamber enclosure approach using LI-COR (c), gas chromatography (d) and IRGA (e); SOC accumulation through chrono-sequence approach using (f) linear- and (g) non-linear regression; CENTURY model (h); BIOME-BGC model (i); isotopic approach (l); CRANTURFC model (m); DAYCENT model (n). Table 2.2 continues in next pages.

NET ECOSYSTEM GAS EXCHANGE				
site	***former land use***	***tipology***	***-NEE***	***pubblication***
Temple (Texas), United States	agricultural land	*Cynodon dactylon*	-1.00 ÷ +8.00 (a)	Dugas, et al., 1999
Minneapolis (Minnesota), United States	unknown	urban lawn (*Poa pratensis, Festuca arundinacea* and *Lolium perenne*)	-0.80 ÷ +0.85 (b)	Hiller, et al., 2011
Wooster (Ohio), United States	unknown	*Festuca arundinacea* and *Lolium perenne* , colonized by fungal endophyte	-9.30 (c) (82 days)	Singh, 2007
Quebec City, Canada	unknown	urban lawns	+0.41 ÷ -5.45 (d) (188 days)	Allaire, et al., 2008
Victoria, Australia	unknown	urban lawn (*Microlaena spp., Festuca spp., Lolium spp.*)	-30.66 ÷ -39.42 (e)(118 days)	Livesley, et al.,2010
Beijing, China	unknown	urban lawn (*Zoysia japonica*)	- (e) (94 days)	Zhou, et al., 2012

SOIL ORGANIC CARBON ACCUMULATION				
site	*former land use*	*tipology*	*Δ SOC*	*pubblication*
Denver (Colorado), United States	agric. land & nat. grass. agric. land & nat. grass.	fairway (0-11.4 cm) green (0-11.4 cm)	+0.9 (g) +1.0 (g)	Qian and Follet, 2002
Palmerston North, New Zeland	none -> artificially created soil profile	green (0-25 cm)	+0.69±0.08 (f)	Huh, et al., 2008
Irvine (California), United States	unknown not relevant	urban lawn sport fields	+1.39±0.15 (f) 0 (f)	Townsend-Small and Czimczik, 2010
Ohio, United States	agricultural land agricultural land	rough (0-15 cm) faiway (0-15 cm)	+2.64±0.08 (g) +3.55±0.06 (g)	Selhorst and Lal, 2011
North Carolina, United States	none ->artificially created soil profile (USGA)	green (0-7.6 cm)	+0.59 (g)	Carley, et al., 2011
Baltimore City (Maryland), United States	various	residential lawns (0-100 cm)	+0.82 (f)	Raciti, et al., 2011
Auburn (Alabama), United States	unknown unknown unknown	residential lawns (0-15 cm) residential lawns (15-30 cm) residential lawns (30-50 cm)	+0.26 (f) +0.11 (f) 0 (f)	Huyler, et al., 2013
8 eco-regions of the United States	unknown	home lawn (0-15 cm)	+0.9 ÷ +5.4 (g)	Selhorst and Lal, 2013

CARBON FLUXES MODELS				
site	***former land use***	***tipology***	***NECB***	***pubblication***
Denver (Colorado), United States	native grassland native grassland	fairway (0-20 cm) green (0-20 cm)	+0.9 ÷ +1.2 (h) +0.6 (h)	Bandaranayake, et al., 2003
Colorado Front Range, (Colorado) United States	agricultural	*Poa pratensis*	+0.62 (h)	Qian, et al., 2003
48 states of the United States	unknown unknown unknown unknown	lawn without inputs high fert. lawn (removing clippings) high fert. lawn (recycling clippings) medium fert. lawn (recycling clippings)	-0.08 ÷ +0.07 (i) +0.27 ÷ +0.60 (i) +0.64 ÷ +1.49 (i) +0.37 ÷ +0.67 (i)	Milesi, et al., 2005
Nebraska City (Nebraska), United States	native praire native praire native praire native praire	*Festuca spp.* (non irr.) *Festuca spp.* (irr.) *Poa pratensis* *Agrostis palustris*	+0.52 (l) +0.74 (l) +0.32 (l) +0.78 (l)	Qian, et al., 2010
United Kingdom	unknown unknown	rough+fairway rough+fairway	+1±0.1 (m) +1±0.1 (m)	Bartlett and James, 2011
Colorado Front Range, (Colorado) United States	agricultural land agricultural land	medium-quality lawn high-quality lawn	+0.78 (n) +1.13 (n)	Zhang, 2012

REMOTE SENSING MODELS				
site	***former land use***	***tipology***	***NPP***	***pubblication***
Falcon Heights-Roseville, (Minnesota) United States	various various	urban turfgrass (low inputs) urban turfgrass (high inputs)	+11.0 +7.71	Wu and Bauer, 2012

2.3.3.1. Net ecosystem gas exchange

Several studies utilized portable photosynthesis systems with chambers, EC, and other techniques to determine CO_2 exchanges between atmosphere and bare soil or low-stature canopies, such as tundra, grassland, forest understory vegetation and various crops. However, few works adopted the same techniques to study the C source/sink potential of turfgrass.

Small surface chambers connected to gas analysers or portable photosynthesis systems are getting increasingly popular to measure CO_2 and NO_2 fluxes. The estimation of NO_2 emissions is particularly relevant because of its higher global warming potential (GWP) compared to the CO_2 (298 times), especially given the usual high fertilization rate used for turfgrass management. Several studies (Pataki, 2006; Livesley, et al., 2010; Townsend-Small and Czimczik, 2010; Zhang, et al., 2013) accounted for N_2O fluxes to determine the best fertilization practices. This kind of works has been recently boosted by the need of high quality turfgrasses with the lower environmental footprint. Several golf clubs in all over the world are participating to environmental programs (e.g. Audubon International, Committed to green, Golf environment Organization) which are pushing towards more sustainable management policies (Wheeler and Nauright, 2006).

Despite the recognised importance of N measurements, accurate determinations of CO_2 fluxes are needed both for eco-physiological investigation and for their crucial role on climate change. In fact the increase in atmospheric CO_2 concentration is the main responsible of the radiative forcing, that represents an index of the natural and anthropogenic drivers of the climate change (IPCC, 2013).

Available data on gas exchange between atmosphere and urban lawns show a wide variability (Table 2.2), changing according to the season in which the experiment has been carried out, the methodology and the latitude. Only few of them however provide annual (Dugas, et al., 1999; Hiller, et al., 2011) or seasonal values of NEE (Allaire, et al., 2008; Livesley, et al., 2010; Zhou, et al., 2012).

The only study that measure NEE using EC system was conducted in a suburb of Minneapolis–Saint Paul, Minnesota (US), in a low-maintenance lawn (such as green area of residential neighbourhoods or a city park) composed by *Poa pratensis*, *Festuca arundinacea* and *Lolium perenne* (Hiller, et al., 2011). The measured fluxes were influenced by nearby CO_2 sources from vehicular road traffic. Traffic effects on measured CO_2 fluxes were estimated through a an empirical model and removed from the C budget, showing annual NEE of +0.80 t C ha^{-1} for 2007 and -0.85 t C ha^{-1} for 2008 (data deduced by Fig.7, Hiller, et al., 2011). Lewis (2010) used EC for testing a custom surface chamber at the Rocky Ford Turfgrass Research Centre and Ashland Bottoms in Kansas (United States). The results confirmed that measures

taken with a closed SC and IRGA (NEE of -12.3 µmol CO_2 m^{-2} s^{-1}) were comparable with measures collected from the EC tower (NEP of 14.4 µmol CO_2 m^{-2} s^{-1}).

Another study (Dugas, et al., 1999) used a micrometeorological method called Bowen ratio/energy balance (BREB) for calculating the flux above canopy of *Cynodon dactylon*. Bowen ratios were calculated from the average temperature and humidity gradients, measured by the BREB instrumentation every 2 seconds at two heights above the canopy. The first year in which the turf was sprigged (planted with stolons) the NEE was +1.00 t C ha^{-1} yr^{-1} (C source), with negative values only in June and July. The second year the increase of the leaf area index (LAI) leaded to a C uptake of -8.00 t C ha^{-1} yr^{-1}, with negative values present from April to October. Maximum daily fluxes in the bermudagrass reached 10.52 µmol CO_2 m^{-2} s^{-1}.

In a more recent study carried out in Canada, Allaire et al. (2008) reported emissions from two urban lawns subjected to different maintenance practices. Carbon dioxide gas exchange was measured using gas chromatography and an acrylic closed chamber, through weekly surveys from May to November. A frequently mowed lawn emitted 5.45 t C ha^{-1} yr^{-1}, while a lawn mowed infrequently emitted 0.41 t C ha^{-1} yr^{-1} in the growing season. The variations observed varied from -1.29 to 14.32 µmol CO_2 m^{-2} s^{-1}. During the studied season NEE resulted always positive in both sites, with emissions close to zero during springtime that increased during the summer due to not limiting soil water content and high temperatures. Furthermore, the study showed that mowing frequency had higher impact on CO_2 flux than fertilisation and soil characteristics.

Also Livesley and colleagues (2010) measured CO_2 fluxes during summer and autumn (from August to November), using an automated chamber system. The measurements were taken in Victoria (Australia) above urban lawns (*Microlaena spp., Festuca spp., Lolium spp.*) treated with different water and nutrient management. They found CO_2 fluxes between 4.6 and 18.7 µmol CO_2 m^{-2} s^{-1} (data deduced by Fig.5, Livesley, et al., 2010) and a cumulative NEE of 30.66 - 39.42 t C ha^{-1} per four months. Irrigated and fertilized turf showed higher emission values during autumn months. The same approach was used by Singh (2007) that estimated CO_2 fluxes in plots of *Festuca arundinacea* and *Lolium perenne* at the Ohio Agricultural and Research development Centre in Wooster, Ohio (United States). Soil surface CO_2 flux was measured between June and September. The CO_2 flux rates resulted higher at the beginning of summer as compared to the end of summer and ranged from 3.15 to 15.15 µmol CO_2 m^{-2} s^{-1}, with an average of 10.10 µmol CO_2 m^{-2} s^{-1} for the entire season, which amounts to a loss of 9.3 t C ha^{-1} in 90 days.

Zhou, et al. (2012) measured the NEE during the transition from winter C source to spring C sink, in a temperate urban lawn (*Zoysia japonica*) in Beijing. The NEE was measured daily with different automatic chambers from January to March and

resulted +0.13 µmol CO_2 m^{-2} s^{-1}. However, results showed that *Zoysia japonica* kept photosynthetic capacity during winter time and activated it when the climate conditions became favourable for CO_2 uptake. Positive correlations were found between NEE and PAR, and NEE and temperature, both well fitting using rectangular hyperbola functions.

2.3.3.2 Soil organic carbon accumulation

The chrono-sequence approach

Considering turf biomass constant over the time, SOC changes are comparable to NEE. Despite SOC dynamics are mainly affected by small-scale ecosystem and cultivation practices (Konen et al. 2002), some efforts have been made to find a general rule describing the dynamic of SOC after turf establishment. Based on the present review of available data regarding SOC variations in American urban lawns and golf courses, a general agreement on the accumulation of SOC in such systems emerged, with rates ranging between 0.11 and 3.55 t C ha^{-1} yr^{-1}.

Qian and Follett (2002), using long-term soil sampling data from 12 golf courses in USA, estimated that total C sequestration continued up to 31 years after turfgrass establishment in fairways and 45 years in putting greens. These latter differ from the previous for both soil texture and turf management, and the authors suggest that higher soil aeration and sand top-dressing may increase the time required to reach the equilibrium of SOM on greens. In fact, SOM accumulation potential increases together with clay content, because of the formation of bonds between the surface of clay particles and OM that retard the decomposition process, therefore the accumulation in a sandy texture soil can result slower.

Although management operation (coring) and high decomposition rate reduce C accumulation on putting green, the intensive water and nutrient management probably promotes relatively high C sequestration rates (Huh, et al., 2008). As suggested by Jastrow, et al. (2007) the avoidance of excessively wet or dry conditions may guarantee on these surfaces the optimal oxygen levels for humification.

Some authors investigating turfgrass (Qian and Follett, 2002; Carley, et al., 2011; Selhorst and Lal, 2011 and 2013), together with other that examine the conversion from agriculture to prairies (Post, et al., 2004), used a non-linear regression model to estimate the sequestration of C over time in turfgrass system. Depending on the environmental variables and the soil typology (structure, texture, etc.), the C sequestration increases for a certain period until the steady state (Figure 2.8).

Other authors however suggested that a linear regression would better describe the process of C accumulation, both for grassland (Lugo, et al., 1986; Conant, et al., 2001; McLauchlan, et al., 2006) and turfgrass (Huh, et al., 2008; Townsend-Small

and Czimczik, 2010; Raciti, et al., 2011; Huyler, et al., 2013). Huh and colleagues (2008) argued that the studies using a non-linear regression don't consider the change of C inputs to the soil over time and therefore the apparent plateau observed in C sequestration rate could be explained by such a decrease in C inputs. Whereas no decrease occurs under putting greens, because SOC pools are altered by opposing factors (such as fertilization, irrigation, coring and mowing) and continue annual sequestration with a linear trend line, avoiding the achievement of a C saturation point.

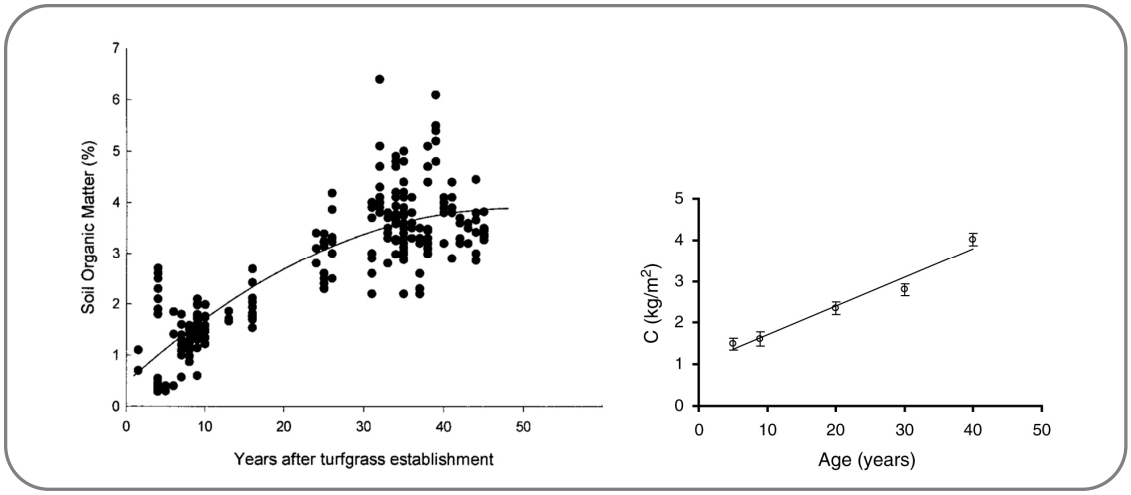


Figure 2.8: The chrono-sequence approach may use non-linear or linear regression for describing soil C trend. Non-linear regression (Qian and Follet, 2002) on the left and linear-regression (Huh, et al., 2008) on the right.

Qian and Follett (2002) estimated total C sequestration rates ranging between 0.9 and 1.0 t ha^{-1} yr^{-1} during the first 25-30 years after turfgrass establishment. Such rates of C accumulation match the findings of Gebhart and colleagues (1994), who reported a C sequestration rate of 1.1 t ha^{-1} yr^{-1} analyzing soil sampling in a temperate cultivated land converted by perennial grassland. Compared with the average accumulation rate of 0.34 t C ha^{-1} yr^{-1} estimated for conversion of cropland to natural unmanaged grassland (Lugo, et al., 1986), the higher sequestration values obtained in golf courses can be attributed to the effect of intensive management practices such as fertilization and irrigation, typical of these systems.

In last decades climate warming has been a new variable that influenced vegetation physiology, with opposing effects on SOC sequestration. The heat and drought stress in some regions decreased the photosynthetic capacity (Zhou, et al., 2012). However, in some regions the increase of temperature have protracted the growing season, permitting ecosystem photosynthesis to start early with a positive effect on NEP (Baldocchi and Wilson, 2001).

Regardless of climate warming, it has been shown that urban lawns soil respiration increase rapidly as soils warm in the spring, with high rate sustained from April until October, but no relation has been found with natural precipitation patterns (Kaye, et al., 2005).

Together with the intensity of maintenance, time since establishment and environmental conditions, C sequestration rates in turfgrass depends also on the previous land use, which affect the percentage of SOM present on site.

Pouyat et al. (2009) report a comparison of SOC stocks between native soil and residential turf grass in two different climatic regions: Baltimore, Maryland, and Denver, Colorado. The native soils of Baltimore (rural forests) showed higher SOC content than shortgrass steppe of Denver. In both cases turfgrass areas established on native soils increased the former SOC content. Despite different starting SOC content, Denver turfgrass had similar SOC content values compared to Baltimore. Qian and Follett (2002) found that fairways that are converted from agricultural land exhibited 24% lower SOM content than fairways converted from native grasslands. Whereas previous land use had little influence in determining SOM in the establishment of putting greens and tees: these playing areas are established at the top of subsequently added sandy substrate, therefore the native soil has no influence.

Intensive agricultural practices lead to the loss of SOC if compared to native grasslands, in which the presence of perennial plants and the lack of tillage allow SOM accumulation (Post and Kwon, 2000; Conant, et al., 2001). The C sequestration potential of agricultural soils that have been allowed to revert to native grassland sites has been proved (Wagai, et al., 1998; McLauchlan, et al., 2006). The conversion of agricultural soils to highly managed systems (such as golf courses, or at least to urban forestry areas) can increase the sequestration rates by the effect of an accelerated C cycling caused by the intensity of management to the presence of a continuous cover of grass (Higby and Bell, 1999).

Generally grasslands (or other ecosystems with high SOM) are close to soil C saturation, hence C sequestration occur slightly. Whereas agricultural soils are C depleted by intensive farming and the establishment of turfgrass can activate C sequestration (Selhorst and Lal, 2011).

In such systems the largest part of the OM is included in the below ground biomass. The above- to below ground biomass ratio in turfgrasses is between 1/9 and 3/7 according to the height of cut, while in forests is 7/3 for hardwoods and 8/2 for conifers (Wood, 2010). However, the rooting depth rarely exceeds 30 cm, concentrating the inputs of C in the shallow soil layers.

Table 2.2 shows the depth of soil samples (when available) of the studies considered in this review. Each method justifies the sampling depth according to the typology of turf, species composition and maintaining intensities.

Analysing the OM content at various interval of soil depth on putting greens in North Carolina (United States), Carley, et al. (2011) found that more than 90% of the C accumulation occurred in the first 7.6 cm. The C accumulation rate, derived from a chrono-sequence considering 196 sampling sites (putting green) in 49 golf courses, resulted 0.59 t C m^{-2} yr^{-1} (0-7.6 cm) over 25 years notwithstanding the periodical disturbance caused by the aeration practices. Moreover, the study of the surface soil layers showed that OM accumulated rapidly near the soil surface (0-2.5 cm) and slowly at higher soil depth. Despite the management influenced the process diluting and removing portions of the accumulating OM, changes in SOM followed a predictable ecological pattern differencing between different depth layers.

In 11 golf courses of central Ohio (United States), Selhorst and Lal (2011) found sequestration rates of 2.64 and 3.55 t C m^{-2} yr^{-1} respectively for roughs and fairways, at 0-15 cm depth. As in the previous study (Carley, et al. 2011), SOC sequestration rates were higher in the top layers (0-5 cm) than in the 5-15 cm layers, confirming the ecological pattern that characterize soils. Despite the same land use (golf courses) the tilling processes on putting greens considerably reduce the C accumulation rate if compared to roughs and fairways (unperturbed turfgrass).

Similarly, a recent study (Selhorst and Lal, 2013) showed that, in conditions of total absence of perturbation, surface layers had higher accumulation rates and sink capacities per unit area than sub-soil layers. The rate of SOC sequestration was 1.2±0.2 t C ha^{-1} yr^{-1} for 0–5 cm depth compared with 0.7±0.1 t C ha^{-1} yr^{-1} for 10-15 cm depth. However Huh et al. (2008) showed that in putting greens C sequestration rate in the first 10 cm was significantly lower than the rate of the underlying 15 cm (0.13±0.04 and 0.57±0.09 t ha^{-1} yr^{-1} respectively). This difference has been associated to the intensive soil management of the superficial horizon, compared to the deeper layers and to the other undisturbed surfaces (fairways and roughs).

Considering studies on residential urban lawns in Alabama (United States), Huyler, et al. (2013), using the chrono-sequences approach, estimated a mean soil C accumulation in the 0-15 cm layer, of 0.26 t C ha^{-1} yr^{-1} which is lower if compared to above cited studies (for example Huh, et al., 2008; Carley, et al. 2011). To explain the low C accumulation the authors postulated that the absence of tree biomass in the studied lawns could play a significant role. In fact, a high percentage of tree roots can develop at shallow horizons under the turfgrass, contributing to C accumulation by fine roots turnover (Huyler, et al., 2013).

Carbon sequestration in turfgrasses has been proved to be strongly related to soil N content. A chrono-sequence study investigating a wider soil depth (0-100 cm) under urban lawns at 32 residential sites around Baltimore, Maryland (US), found that turfs have the potential to accumulate C (0.82 t C ha^{-1} yr^{-1} in average) and N (0.083 t N ha^{-1} yr^{-1} in average) (Raciti, et al., 2011). Rates of N accumulation are similar in magnitude to estimated fertilizer N inputs, confirming the high capacity for N retention of residential lawns. The strong positive relationship between C and N suggests that C may be accumulating along with N inputs. Management practices (N fertilization, clippings return) that increase lawn productivity enhance soil C sequestration more than OM decomposition (Townsend-Small and Czimczik 2010; Raciti, et al., 2011). On the other hand, lawns could become a strong source of GHGs emissions under high N fertilizer inputs (Bremer, 2006; Groffman, et al., 2009; Livesley, et al., 2010).

Isotopic approach

Another method to assess SOC dynamics in turfgrass systems and more widely in all terrestrial ecosystems is the isotopic approach. Qian and colleagues (2010), using isotope techniques, founds that 4 years after establishment of different turfgrass species, from 9 to 14% of SOC of the 0-20 cm soil layer was derived from turfgrass. All turfgrasses exhibited significant C inputs, varying from 1.39 to 3.35 t ha^{-1} yr^{-1} (in Fine fescue rain fed and irrigated). C sequestration rates varied from 0.32 to 0.78 t C ha^{-1} yr^{-1} depending on species. Despite irrigation increased the net OC input to the 0-20 cm soil profile by 141%, at the same time also increased SOC decomposition by twofold.

2.3.3.3 Carbon fluxes models

Carbon fluxes of urban vegetated areas can be estimated using ecosystem process models. Modelling provides a way for scientists to enlarge their research in situations where field measurements are difficult, either because of the number of different treatments to consider or because of the large scale to cover.

Ecosystem models

Ecosystem process models simulate C, water, and N cycles. Most of these models have been developed for natural ecosystems and consider fluxes deriving from biological processes. However, several studies have successfully applied models such CENTURY, DAYCENT, CRANTURFC and BIOME-BGC to urban systems, taking into account also natural activities variations driven by human processes. Carbon fluxes can be divided in vertical and horizontal: vertical fluxes involve plant-soil systems and include photosynthesis, respiration and anthropogenic emissions, while horizontal fluxes are mostly driven by human activities and include flows of trash and material. The ecosystem process models are able to estimate only vertical C fluxes and to simulate responses of vegetation in the cities and in the urban footprint to the increased atmospheric CO_2 concentration caused by urban pollution, and to the changes in urban climate.

Several typical management characteristics need to be considered when modelling golf course turfgrass compared to other ecosystems. Practices, such as the removal of the shoots and the return as clippings during the frequent cuts, the occasional aerification, the presence of a layer of dead organic matter generated by the residues (thatch), the intensive management consisting in irrigation and fertilization, etc., are processes that can generate multiple and contrasting effects on C fluxes, making the parameterization of the model very difficult.

The CENTURY model, developed in the United States to evaluate C dynamics in grasslands (Parton, et al., 1987), has been parameterized and used to simulate turfgrass ecosystems in golf course and home lawn conditions (Bandaranayake, et al., 2003; Qian, et al., 2003). The CENTURY model estimates long-term SOC trends based on mathematical representation of C-cycling processes in the soil-plant-system, at monthly time-step. It is adequate for simulations of medium- to long-term changes in SOC, plant productivity, and other ecosystem parameters.

Bandaranayake et al. (2003) evaluated the long-term SOC dynamics in turfgrass using CENTURY. The major input variables included climate and soil data, lignin content of plant material, plant tissue C to N ratio, initial soil C and N and soil N inputs through fertilization (Bandaranayake, et al., 2003). The model simulated the conversion from shortgrass prairie to fairway showing an increase in SOC. Despite the complexity, the simulation indicated that turfgrass can act as a C sink, giving back results in line with other studies based on direct measurements of SOC

changes. The model estimated an accumulation of 23 to 32 t ha^{-1} of C in the 0 to 20 cm soil layer in fairways after 30 years. The average rate of C accumulation in the three decades since the establishment of turf ranged from 0.9 to 1.2 t ha^{-1} yr^{-1} and the time needed to reach a relatively steady state on SOC content depended on soil texture, confirming the important role of clay particles in providing greater protection to SOM compared to sand and silt.

Qian and colleagues (2003) evaluated the ability and the effectiveness of the same model to predict SOC changes overtime in home lawns (*Poa pratensis*), in relation to the returning of turf-clippings to the ground after mowing. CENTURY model tested against field measurements correctly simulated the annual cumulative clipping yields over three years, and it simulated an average C accumulation rate of 0.62 t ha^{-1} yr^{-1} over 50 years under a clippings-returning scenario coupled to high N fertilization.

The DAYCENT model (Parton, et al., 1998) is the more recently developed, daily time-step version of the CENTURY model. It uses a finer time scale than CENTURY in modelling decomposition, nutrient flows, soil water, and soil temperature, and has an increased spatial resolution for soil layers. The model has been applied to several agro-ecosystems to simulate and predict daily carbon and nitrogen dynamics (Del Grosso, et al., 2006; Stehfest, et al., 2007; Del Grosso, et al., 2009). Zhang (2012) parameterized this model with field data from *Poa pratensis* in order to modify and validate it for lawns, with the aim to predict the long-term impact of different management practices on primary productivity and C sequestration.

Based on the assumption that soil C sequestration is function of plant productivity and SOM decomposition (Parton et al., 1987), in Zhang simulation (2012) SOC increased with increasing irrigation rate and partially with increasing fertilization rates. DAYCENT simulated an higher C sequestration rate in a highly managed lawn compared to a medium-quality lawn, with 1.13 t C ha^{-1} yr^{-1}) and 0.78 t C ha^{-1} $yr^{-1,}$ respectively in the first 10 years after conversion from agricultural land (Zhang, 2012).

Using the ecosystem model BIOME-BGC, Milesi, et al. (2005) simulated the growth of turf in different climatic zones of the United States, predicting the accumulation of C under different scenarios of maintenance, including either removal or recycling of the grass clippings, different N fertilization rates and alternative water irrigation practices. Carbon accumulation rate ranged from -0.08 to +1.49 t C ha^{-1} y^{-1}. The comparison between different scenarios showed a positive NECB. The study indicates that C sequestration in the system increases with the availability of N.

LCA model

Bartlett and James (2011) adopted a modelling approach based upon a LCA expressly developed for golf course turfgrass, called CRANTURFC. The model is based on a mass balance equation, given by the algebraic sum of CO_2 sequestered by the plant-soil system and the CO_2eq emissions from maintenance of turfgrass (Figure 2.9).

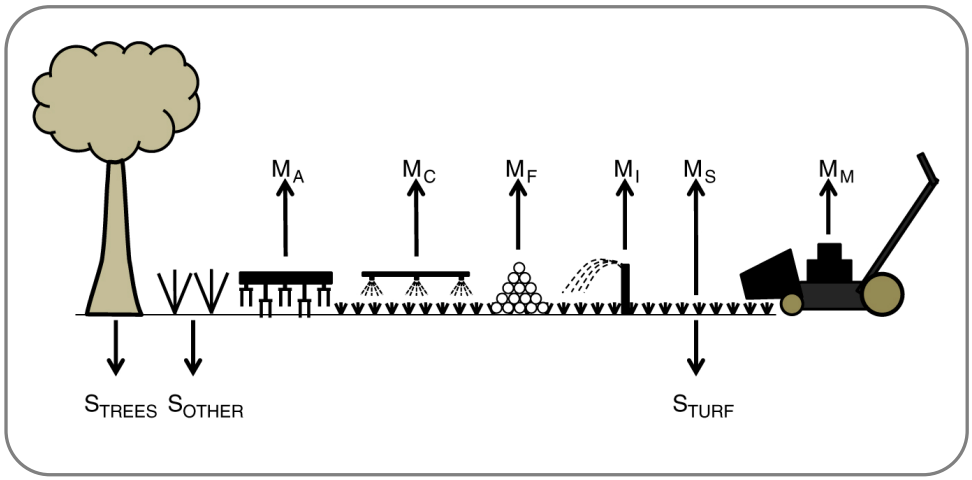


Figure 2.9: The CRANTURFC greenhouse gases (GHGs) balance model illustrating components within the system boundary for a golf course (Bartlett and James, 2011).

The GHGs efflux from maintenance is the sum of emissions from agrochemical and fertiliser application, irrigation application, mowing, aeration and soil basal respiration. The model takes in account also GHGs emissions from the decomposition of grass clippings.

The study collected input data derived from published works from agriculture and turfgrass management, and applied the model to two British golf courses. The balance resulted to be negative for the intensively managed greens and tees, which acted as CO_2 sources, compared to fairways and roughs that were found to be C sinks. The most marked component of the emissions from fairways and roughs was mowing, while greens and tee emissions came from the application of N fertiliser. On the whole, playing areas (turfgrass) were a sink of C (-0.54±0.11 t C ha^{-1} yr^{-1}), as fairways and roughs balanced the emissions of greens and tees. The C sequestration of turf amounted to +1±0.1 t C ha^{-1} yr^{-1}. Other vegetation areas of the course provided net sequestration, particularly areas with trees (not-playing components), which exhibited sequestration rates 2.8 times greater than turfgrass.

Remote sensing models

Gross and net primary productivity of urban turfgrass can be also estimated with models based on remotely sensed data. Remote sensing models are useful instruments for mapping urban ecosystem services and quantifying above ground C storage (Davies, et al., 2011). Remotely sensed imagery provides us a synoptic view of the Earth's environment without intensive labour and exhaustive field surveys.

Through the analysis of satellite images and the development of models is possible to identify land uses and to estimate productivity indexes such as GPP and NPP (Franklin, 2001). However, the estimation of C budgets becomes challenging in urban and suburban areas because of the high spatial heterogeneity of urban landscapes. Although traditional urban land use already includes categories such as residential areas, commercial areas, etc., those definitions don't provide details about vegetation types and characteristics, while high resolution imagery can detect fine details and turfgrass, for instance, can be accounted for. However, the amount of shadows existing in the imagery may underestimate the turfgrass surface and consequently the C fixation. This limitation can be fixed investigating spectral reflectance of shadows, detecting these surfaces and including them into the amount of turfgrass area (Wu and Bauer, 2012).

Several production efficiency models based on remote sensed data have been employed to quantify C budgets of natural ecosystems and croplands. Wu and Bauer (2012) developed a specific model to estimate annual NPP of turfgrass, using field biophysical measurements to parameterize the fraction of absorbed photosynthetically active radiation (fAPAR). Their work showed that NPP of turfgrass varied significantly across the studied area due to different categories of management practices. The categories were identified comparing field data and values of the Normalized Difference Vegetation Index (NDVI) obtained from the analysis of Quick Bird images. High input turfgrass (e.g., golf course grass) had high average NPP (11.0 t C ha^{-1} yr^{-1}) while the productivity of low input turfgrass (e.g., roadside turfgrass) was relatively low (7.712 t C ha^{-1} yr^{-1}).

2.4 Discussion

The majority of the studies considered in this review deals with golf courses and residential-urban lawns (7⁄18 and 11⁄18 respectively). Although data regarding C sequestration of lawns are more abundant, studies on golf courses generally predict more precisely C sequestration rates as the uniformity of their surfaces is very high.

Three quarters of the studies were carried out in USA and included direct measurements (net ecosystem gas exchange and SOC accumulation) and use of models. The development and improvement of new methodologies demonstrate the increasing interest for this topic. The high presence of technical turfgrass in the USA and the need to reach higher quality turfgrasses probably have been boosting research on this country. A small number of studies were carried out in Australia (Huh, et al., 2008; Livesley, et al., 2010), Europe (Bartlett and James, 2011) and China (Zhou, et al., 2012, was the only available study, but more researches have been conducted). The lack of studies on C cycling in technical turfgrasses in Europe

and in the Mediterranean area is surprising, if compared to the wide presence of such systems (See Chapter 3).

2.4.1 Comparison of methodologies

Only few studies estimated turfgrass NEE (Table 2.2) using EC, because micro-meteorological approach is very expensive and requires high uniformity of surfaces over big areas, resulting difficult to apply in systems such golf courses or urban lawns because they are often enclosed between trees and buildings or affected by disturbance of factors such as road traffic (Hiller, et al., 2011).

Small surface chambers measurements systems, on the other hand, are cheaper, easier to install and operate, often portable and they allow working on multiples treatments and/or multiple gases in smaller areas (Pataki, 2006; Livesley, et al., 2010).

The alteration of environmental parameters inside the chamber (CO_2 concentration, pressure, temperature, etc.) can result in methodological errors affecting the measurements. The variability among measurements and the difficulty to collect measurements for long periods make complicated the integration of data at seasonal or annual scale. As a consequence NEE measurements with SC often refer only to daily (See Chapter 3) or seasonal surveys, concentrated in the turfgrass growing season (from spring till autumn) (Allaire, et al., 2008; Livesley, et al., 2010; Zhou, et al., 2012), neglecting dormancy season and not providing annual estimation.

The utilization of automatized chamber can certainly decrease the labour for the field measurements and lead to more complete data sets, permitting nightly and long-term surveys.

Direct SOC sequestration assessment by soil analysis gives more easily annual estimates of net C fluxes for turfgrass (where the above ground NPP can be considered as at steady state). At the same time also measurement of C isotopic composition (Qian, et al., 2010) appears to be an appropriate approach to study SOC dynamics. Nevertheless, it is critical to predict SOC trends for soil-geographic regions being SOC dynamics affected by several factors, such as temperature, moisture, and microorganism activity. The facility to collect SOC data makes the chrono-sequence method widely utilized in the literature (Qian and Follet, 2002; Huh, et al., 2008; Townsend-Small and Czimczik, 2010; Selhorst and Lal, 2011; Carley, et al., 2011; Raciti, et al., 2011; Huyler, et al., 2013; Selhorst and Lal, 2013) however the adoption of linear or non-linear regression models in determining the behaviour of C sequestration over time decisively influences the estimates of soil C accumulation rates.

Remote sensing models are useful instruments for quantifying above ground C storage. However, neglecting below ground C pools and fossil fuel consumption, they cannot explain the whole C budget of the system. The relation between field measurements and images vegetation index (e.g. NDVI) provide good results that depend on vegetative conditions and management intensities (Wu and Bauer, 2012). The estimate of the quantities and spatial patterns of the annual NPP of turfgrass can better represent the importance of these areas for climate change, accounting for the actual C stocks.

Several models developed for other ecosystems have been used also for turfgrasses with results in agreement with the studies that used direct measurements of SOC sequestration. As an example, models outputs showed that highly managed lawns have higher C sequestration rate than a medium-quality ones (Milesi, et al., 2005; Zhang, 2012), confirming the strong relationship between soil C sequestration and plant productivity and SOM decomposition (Parton et al., 1987).

The development and the upgrade of ecosystemic models helped to understand the behaviour of turfgrasses, permitting the study of different treatments (management) without excessive costs (Bandaranayake, et al., 2003; Qian, et al., 2003; Zhang, 2012). However, the development of specific models for turfgrass (Bartlett and James, 2011) that considers all the components provide more detailed and complete results.

Data of Table 2.2 show that measurements system affect the estimates of C balance. In fact, all the studies adopting SC measurements systems provide negative results, suggesting a general underestimation of C sequestration related to that approach. On the contrary, models and chrono-sequences approaches lead to positive C sequestration, while the small amount of studies using micrometeorological system does not provide enough evidence to outline a trend.

2.4.2. Factors driving the C sequestration potential of turfgrass

Table 2.3 shows how the mix of variables, such as species composition, environmental conditions, site characteristics, former land use and turf management influence C sequestration in turfgrass ecosystems based on the works considered in the present study.

Studies adopting SC approaches generally reported that the system is a C source, demonstrating that in cool-season grass, decomposition and respiration prevail on photosynthesis during growing season (Bremer and Ham, 2005; Allaire, et al., 2008; Livesley, et al., 2010; Singh, 2007). However other studies, considering only daily surveys (Pataki, 2006; Lewis, 2010; Acuna, et al., 2012), suggest that cool-season grass may act as a C sink (data not showed). Whereas all the data providing from chrono-sequence approaches and modelling systems show positive C sequestration rate, affected from several variables (Table 2.3).

Specific composition seems to have a role in the seasonal behaviour of turfgrass. Few studies on warm-season grass turfs show that daily CO_2 fluxes are near zero in autumn and spring, and then reach a maximum in the summer, due to the high productivity of C4 metabolism (Dugas, et al., 1999; Lewis, 2010). Zhou and colleagues (2012) observed a C source behaviour on turfs during the transition period from winter to spring, however during the days with favourable climate conditions, the activation of the photosynthetic path caused CO_2 uptake to prevail.

	Variables	*Carbon sequestration*
Species composition	C4 metabolism (SUMMER)	↑
	C4 metabolism (SPR & AUT)	↔
	C3 metabolism (SUMMER)	↓
	C3 metabolism (SPR & AUT)	↑
	C4 & C3 metabolism (WINTER)	↔
Environmental condition	Lenght of the growing season	↑
	High mean annual temperatures	↓
	Atmospheric CO_2 concentration	↑
Site characteristic	Depth of the roots	↑
	Soil depth	↓
	Clay texture soil	↑
Former land use	Natural previous land use	↓
	Agricultural previous land use	↑
	Age of turfgrass	↓
Management	Irrigation	↑
	Fertilization	↑
	Release of clippings	↑
	Mowing Frequency	↑
	Shallow aeration (coring)	↓

Table 2.3: Variables influencing C sequestration in turfgrass: increase (↓), decrease (↑) and non-variation (↔).

Generally, fluxes are affected by leaf area, radiation, temperature and soil water content. The rate of C sequestration increase with the increase of these variables (Dugas, et al., 1999; Zhou, et al. 2012). In cool-season grasses turfs, emissions are close to zero during springtime and increase during the summer (Singh, 2007; Allaire, et al., 2008; Livesley, et al., 2010). This can be explained by different mechanisms. First of all, soil respiration has been shown to increase exponentially with increasing temperatures that characterize summer (Schlesinger and Andrews, 2000). In addition, turfgrasses are continuously watered in warm periods in order to avoid drought stress. Water availability however, maintains aesthetic characteristics without affecting the productivity of cool-season turfs during the summer, while soil moisture content influences microorganism activity and soil surface CO_2 flux (Wagai, et al., 1998), further increasing soil respiration and SOC decomposition (Qian, et al., 2010). The negative relationship between SOC and mean annual temperatures highlighted by some authors (Kaye, et al., 2005; Huyler, et al., 2013) may be therefore explained by such mechanism.

Carbon sequestration is related to the efficiency of the ecosystem. Several environmental conditions, affecting photosynthesis and respiration activities, determine the NEP of each site and the amount of C allocated belowground. Net primary productivity for C3 grasses is higher in spring and remains positive until July, whereas C4 plants are the most active herbaceous species in nonurban systems at this time of year.

Soil structure and texture affect SOC accumulation: the presence of clay retards the decomposition process (Qian and Follet, 2002; Bandaranayake, et al., 2003), holding C stock. Moreover, the right combination between texture and species is determinant. For example, several herbaceous species which are native from sandy soils grow up better and import more C as compared to loamy soils (Huyler, et al., 2013). Regardless of the edaphic conditions, the quantities of below ground biomass produced by each species have not been investigated yet.

Also the SOC content, which depends both on the previous land use and the time since turf establishment, influence the C sequestration rate. Low SOC content allow higher rate of C sequestration (Qian and Follet, 2002; Pouyat, et al., 2009; Selhorst and Lal 2011), in agreement with the soil carbon saturation concept (Stewart, et al., 2007). According to this theory, each soil have a maximum amount of C stabilized (saturation limit, steady state), therefore SOC accumulation will decrease at higher SOC content (non-linear SOC accumulation).

Natural undisturbed sites (such as grasslands) generally have higher SOC content than agricultural lands and turfgrasses established in the former report lower C sequestration rate compared to that one established the latter (Qian and Follet, 2002; Pouyat, et al., 2009; Selhorst and Lal 2011). Different turfgrass systems have

the possibility to reach similar SOC densities, despite regional variations in climate, parent material, and topography. Management efforts can offset the constraint of climate and of native land use, indicating that anthropogenic factors may overwhelm native environmental factors in controlling SOC (Pouyat, et al., 2009).

Irrigation has a different effect on SOC dynamics depending on the season and on the interaction with N fertilization. In spring and autumn irrigation and fertilization enhance plant productivity, increasing C sequestration rates in plants and C accumulation in soil (Livesley, et al., 2010; Milesi, et al., 2005; Raciti, et al., 2011). The strong positive relationship between C and N suggests that C may be accumulating along with N inputs. Nitrogen inputs (fertilization and/or clippings return) that increase lawn productivity enhance soil C sequestration more strongly than they enhance decomposition. On the other hand lawns could become a strong source of GHGs emissions under high N fertilizer inputs. For this reason the release on field of turf clippings during mowing is an effective strategy to enhance SOC sequestration (Qian, et al., 2003; Milesi, et al., 2005) without affecting N_2O emissions.

Other maintenance operations play a crucial role on C sequestration of turfgrasses. Regular mowing frequency does not place any particular stress on turfgrass plants (Hull, 2000). However higher mowing frequency has shown to have a impact on CO_2 flux, increasing NEP (Allaire, et al., 2008).

Soil aeration by coring, generally carried out only on putting greens, removes OC on the shallow layers of soil, together with the thatch. The removal of OM and the substitution with sand (top-dressing) both decrease SOC content and increase its bioavailability for microorganisms in the upper layer, enhancing the CO_2 flux to the atmosphere (Qian and Follett, 2002).

High managed surfaces such as greens have high C sequestration rate, thanks to fertilization and irrigation (Huh, et al., 2008; Carley, et al., 2011). Moreover, the major mechanism of soil C sequestration in the sandy soil under a putting green has shown to be the humification (Jastrow et al. 2007; Huh, et al., 2008), which reduces the bioavailability of soil C. However soil C accumulation mainly occurred in the undisturbed part of the topsoil, from 10 to 25 cm depth in greens, where the stabilization processes are not disturbed (Huh, et al., 2008; Carley, et al. 2011; Selhorst and Lal, 2011 and 2013). Carbon sequestration in greens depends on the construction and management operations: establishment with sandy soils, irrigation and fertilization enhance C accumulation, while periodical aeration reduces it. Regardless of coring, the rate of C sequestration, and hence the time to reach equilibrium, decreased with increase in soil depth (Selhorst and Lal, 2011 and 2013).

Despite greens receive more inputs than fairways and roughs, the decrease of SOC and the increase of C bioavailability due to the coring operations may be the key

reason of the higher SOC values present in the latter (0.9-3.55 t C $ha^{-1}yr^{-1}$ and 0.69-1.2 t C $ha^{-1}yr^{-1}$ respectively) (Qian and Follet, 2002; Bandaranayake, et al., 2003; Huh, et al., 2008; Selhorst and Lal, 2011; Carley, et al., 2011; Bartlett and James, 2011).

This review doesn't consider sport fields turfgrass, which are far more artificial systems than the ones considered above. In sports fields there is no C sequestration over time because of the frequent surface restoration. Athletic fields are renovated extensively every 5-10 years, including tilling and re-sodding to replace dead grass and frequent aeration practices to offset compaction. Similar practices are employed in conventional agriculture, which disrupts soil C accumulation (Townsend-Small and Czimczik 2010).

2.5 Conclusion

In urbanized landscapes anthropogenic drivers control turfgrass SOC dynamics and may dominate over natural and native drivers. The change in SOC occurring during the urbanization of a site can be fundamental to understand the ecosystem response to urban land-use change, especially moving to turfgrass.

In the last decades a number of approaches and methodologies have been applied to the study of C dynamics in turfgrass system, and despite the scarce comparability among them, some general rules have been outlined by the present study. Turfgrass typology, species composition, fertilization, irrigation, mowing, coring, seasonality, climate, soil texture and previous land use resulted the most important factors affecting SOC sequestration on turfgrass. The CO_2 emission or uptake from turf has clear drivers connected to these variables, while some processes still necessitate of further researches.

We can state that high temperatures accelerate the decomposition of SOC only when soil moisture is adequate, and inhibit decomposition when soil moisture becomes limited.

Generally the studies investigating SOC dynamics show that turfgrasses act as a C sink, both in golf courses and urban lawns, while studies , utilizing SC approach for extended periods, show a C source behaviour, even during the growing season. More investigation need to be conducted, especially to explore limits and advantages of opposite methodologies. The lack of continuous field-measured data is often compensated by the use of models parameterized on the basis of punctual data.

This review wants to represent a basis for future studies, highlighting the need for clarification on terms and for an improvement of the methodologies used to estimate the C sequestration potential of turfgrass.

2.6 Bibliography

Aber J.D. and Melillo J.M. 2001. Terrestrial ecosystems. Academic Press San Diego, pp. 556.

Acuna A., Villalobos L. and Pastenes C. 2012. Carbon sequestration in selected turfgrass species grown in central Chile: preliminay results. 3rd European Turfgrass Conference, Bioforsk Fokus, pp. 33-34.

Alberti M. 1996. Measuring urban sustainability. Environ Impact Asses Rev 996: 38l-424.

Allaire S.E., Dufour-L'Arrivée C., Lafond J.A., Lalancette R. and Brodeur J. 2008. Carbon dioxide emissions by urban turfgrass areas. Canadian Journal of Soil Science 88: 529-532.

Baldocchi D.D. and Wilson K.B. 2001. Modeling CO_2 and water vapor exchange of a temperate broadleaved forest across hourly to decadal time scales. Ecological Modelling 142: 155-184.

Bandaranayake W., Qian Y., Parton W., Ojima D. and Follett R. 2003. Estimation of soil organic carbon changes in turfgrass systems using the CENTURY model. Agronomy Journal 95: 558-563.

Bartlett M.D. and James I.T. 2011. A model of greenhouse gas emissions from the management of turf on two golf courses. Science of the total environment 409: 1357-1367.

Beard J.B. 2002. Turfgrass management for golf courses, Ann Harbor Press, pp. 793.

Bremer D.J. 2006. Nitrous Oxide Fluxes in Turfgrass. Journal of environmental quality 35: 1678-1685.

Bremer D.J. and Ham J.M. 2005. Measurement and partitioning of in situ carbon dioxide fluxes in turfgrasses using a pressurized chamber. Agronomy Journal 97: 627-632.

Bullock P. and Gregory P.J. 2009. Soils in the urban environment, pp. 174.

Carley D.S., Goodman D., Sermons S., Shi W., Bowman D., Miller G. and Rufty T. 2011. Soil organic matter accumulation in creeping bentgrass greens: a chronosequence with implications for management and carbon sequestration. Agronomy Journal 103: 604-610.

Cereti C. 2002. Tappeti erbosi e inerbimenti. Baldoni R., Giardini L.: Coltivazioni erbacee–Foraggere e tappeti erbosi, Patron Editore, pp. 354-400.

Chapin III F. S., Woodwell G., Randerson J.T., Rastetter E.B., Lovett G., Baldocchi D.D., Clark D., Harmon M.E., Schimel D.S. and Valentini R. 2006. Reconciling carbon-cycle concepts, terminology, and methods. Ecosystems 9: 1041-1050.

Chapin III F.S. and Matson P.P.A. 2011. Principles of terrestrial ecosystem ecology, Springer, pp. 529.

Churkina G. 2008. Modeling the carbon cycle of urban systems. Ecological Modelling 216: 107-113.

Colangelo G., Davies C., Lafortezza R. and Sanesi G. 2006. L'esperienza delle Community Forests in Inghilterra. Ri-Vista. Ricerche per la Progettazione del Paesaggio (on line) 6: 82-92.

Conant R.T., Paustian K. and Elliott E.T. 2001. Grassland management and conversion into grassland: effects on soil carbon. Ecological Applications 11: 343-355.

Cramer M. 1993. Urban Renewal Restoring the Vision of Olmsted and Vaux in Central Park's Woodlands. Ecological Restoration 11: 106-116.

Croce P., De Luca A., Falcinelli M., Modestini F.S. and Veronesi F. 2006. Tappeti erbosi, Edagricole, pp. 340.

Croce P., Volterrani M., Beard J., De Luca A. and Mocioni M. 2003. Adaptability of warmseason turfgrass species and cultivars in a mediterranean climate. I International Conference on Turfgrass Management and Science for Sports Fields 661, pp. 365-368.

Davies Z.G., Edmondson J.L., Heinemeyer A., Leake J.R. and Gaston K.J. 2011. Mapping an urban ecosystem service: quantifying above-ground carbon storage at a city-wide scale. Journal of Applied Ecology 48: 1125-1134.

De Luca A., Volterrani M., Gaetani M., Grossi N., Croce P., Mocioni M., Lulli F. and Magni S. 2008. Warm-season turfgrass adaptation in northern Italy. Proceedings of the 1st European Turfgrass Society Conference. Pisa (Italy), pp. 75-76.

Decker E.H., Elliott S., Smith F.A., Blake D.R. and Rowland F.S. 2003. Energy and material flow through the urban ecosystem. Annual Review of Energy and the Environment 25: 685-740.

Del Grosso S.J., Ojima D.S., Parton W.J., Stehfest E., Heistemann M., DeAngelo B. and Rose S. 2009. Global scale DAYCENT model analysis of greenhouse gas emissions and mitigation strategies for cropped soils. Global and Planetary Change 67: 44-50.

Del Grosso S.J., Parton W.J., Mosier A.R., Walsh M.K., Ojima D.S. and Thornton P.E. 2006. DAYCENT national-scale simulations of nitrous oxide emissions from cropped soils in the United States. Journal of Environmental Quality 35: 1451-1460.

Dugas W.A., Heuer M.L. and Mayeux H.S. 1999. Carbon dioxide fluxes over bermudagrass, native prairie, and sorghum. Agricultural and Forest Meteorology 93: 121-139.

Eriksson, 2012. Info-graphics, client: Bro Hof Slott Golf Club (2012). "The construction of USGA greens". URL consulted on 2014 February 04. http://www.rithuset.com/illustration/The%20construction%20of%20USGA%20greens.jpg

Escobedo F., Varela S., Zhao M., Wagner J.E. and Zipperer W. 2010. Analyzing the efficacy of subtropical urban forests in offsetting carbon emissions from cities. Environmental Science & Policy 13: 362-372.

Escobedo F.J., Kroeger T. and Wagner J.E. 2011. Urban forests and pollution mitigation: analyzing ecosystem services and disservices. Environmental Pollution 159: 2078-2087.

Franklin S.E. 2001. Modeling forest net primary productivity with reduced uncertainty by remote sensing of cover type and leaf area index. Spatial uncertainty in ecology, pp: 284-307.

Fuller R.A. and Gaston K.J. 2009. The scaling of green space coverage in European cities. Biology Letters 5: 352-355.

Gebhart D.L., Johnson H.B., Mayeux H. and Polley H. 1994. The CRP increases soil organic carbon. Journal of Soil and Water Conservation 49: 488-492.

Golubiewski N.E. 2006. Urbanization increases grassland carbon pools: Effects of landscaping in Colorado's front range. Ecological Applications 16: 555-571.

Groffman P.M., Williams C.O., Pouyat R.V., Band L.E. and Yesilonis I.D. 2009. Nitrate leaching and nitrous oxide flux in urban forests and grasslands. Journal of environmental quality 38: 1848-1860.

Higby J.R. and Bell P.F. 1999. Low soil nitrate levels from golf course fairways related to organic matter sink for nitrogen. Communications in Soil Science & Plant Analysis 30: 573-588.

Hiller R.V., McFadden J.P. and Kljun N. 2011. Interpreting CO_2 fluxes over a suburban lawn: The influence of traffic emissions. Boundary-layer meteorology 138: 215-230.

Hollis J. 1991. The classification of soils in urban areas. Soils in the urban environment, pp. 5-27.

Huh K.Y., Deurer M., Sivakumaran S., McAuliffe K. and Bolan N.S. 2008. Carbon sequestration in urban landscapes: the example of a turfgrass system in New Zealand. Soil Research 46: 610-616.

Hull R.J. 2000. Mowing, its impact on turfgrass. Turfgrass Trend, pp. 1-8.

Hunt S. 2003. Measurements of photosynthesis and respiration in plants. Physiologia Plantarum 117: 314-325.

Huyler A., Chappelka A.H., Prior S.A. and Somers G.L. 2013. Drivers of soil carbon in residential 'pure lawns' in Auburn, Alabama. Urban Ecosystems, pp. 1-15.

ISTAT. 2012. Dati ambientali nelle città: Qualità dell'ambiente urbano.

Jastrow J.D., Amonette J.E. and Bailey V.L. 2007. Mechanisms controlling soil carbon turnover and their potential application for enhancing carbon sequestration. Climatic Change 80: 5-23.

Jim C.Y. 1998. Soil characteristics and management in an urban park in Hong Kong. Environmental management 22: 683-695.

Jo H.K. and McPherson G.E. 1995. Carbon storage and flux in urban residential greenspace. Journal of Environmental Management 45: 109-133.

Kaye J., McCulley R. and Burke I. 2005. Carbon fluxes, nitrogen cycling, and soil microbial communities in adjacent urban, native and agricultural ecosystems. Global Change Biology 11: 575-587.

Kaye J.P., Groffman P.M., Grimm N.B., Baker L.A. and Pouyat R.V. 2006. A distinct urban biogeochemistry? Trends in Ecology & Evolution 21: 192-199.

Konen M.E., Jacobs P.M., Burras C.L., Talaga B.J. and Mason J.A. 2002. Equations for predicting soil organic carbon using loss-on-ignition for north central US soils. Soil Science Society of America Journal 66: 1878-1881.

Konijnendijk C.C., Ricard R.M., Kenney A. and Randrup T.B. 2006. Defining urban forestry–A comparative perspective of North America and Europe. Urban Forestry & Urban Greening 4: 93-103.

Krull E.S., Baldock J.A. and Skjemstad J.O. 2003. Importance of mechanisms and processes of the stabilisation of soil organic matter for modelling carbon turnover. Functional Plant Biology 30: 207-222.

Kutzbach L., Schneider J., Sachs T., Giebels M., Nykänen H., Shurpali N.J., Martikainen P.J., Alm J. and Wilmking M. 2007. CO_2 flux determination by closed-chamber methods can be seriously biased by inappropriate application of linear regression. Biogeosciences 4: 1005-1025.

Lal R., Follett R., Kimble J. and Cole C. 1999. Managing US cropland to sequester carbon in soil. Journal of Soil and Water Conservation 54: 374-381.

Lewis J.D. 2010. Carbon, nitrogen and water fluxes from turfgrass ecosystems. Department of Horticulture, Forestry, and Recreation Resources-College of Agriculture. Kansas State University, Manhattan, Kansas, pp. 155.

Livesley S.J., Dougherty B.J., Smith A.J., Navaud D., Wylie L.J. and Arndt S.K. 2010. Soil-atmosphere exchange of carbon dioxide, methane and nitrous oxide in urban garden systems: impact of irrigation, fertiliser and mulch. Urban Ecosystems 13: 273-293.

Lugo A.E., Sanchez M.J. and Brown S. 1986. Land use and organic carbon content of some subtropical soils. Plant and Soil 96: 185-196.

McLauchlan K.K., Hobbie S.E. and Post W.M. 2006. Conversion from agriculture to grassland builds soil organic matter on decadal timescales. Ecological Applications 16: 143-153.

McPherson E.G., Nowak D., Heisler G., Grimmond S., Souch C., Grant R. and Rowntree R. 1997. Quantifying urban forest structure, function, and value: the Chicago Urban Forest Climate Project. Urban ecosystems 1: 49-61.

McPherson E.G., Nowak D.J. and Rowntree R.A. 1994. Chicago's urban forest ecosystem: results of the Chicago Urban Forest Climate Project. US Department of Agriculture, Forest Service, Northeastern Forest Experiment Station Radnor, PA, pp. 94.

McPherson E.G. and Simpson J.R. 1999. Carbon Dioxide Reduction Through Urban Forestry. Gen. Tech. Rep. PSW-171, USDA For. Serv., Pacific Southwest Research Station, Albany, CA, pp. 237.

Miele S., Volterrani M. and Grossi N. 2000. Warm season turfgrasses: results of a five-year study in Tuscany. Agricoltura Mediterranea 130: 196-202.

Milesi C., Elvidge C., Dietz J., Tuttle B., Nemani R. and Running S. 2005a. A strategy for mapping and modeling the ecological effects of US lawns. 3rd international symposium remote sensing and data fusion over urban areas (URBAN 2005), 5th international symposium remote sensing of urban areas (URS 2005), Tempe, AZ, USA.

Milesi C., Running S.W., Elvidge C.D., Dietz J.B., Tuttle B.T. and Nemani R.R. 2005b. Mapping and modeling the biogeochemical cycling of turf grasses in the United States. Environmental Management 36: 426-438.

Miller R. 1997. Urban forestry: planning and managing urban greenspaces. Waveland Press, pp. 512.

Murphy J.T. 2007. Patterns of carbon dioxide and water vapor flux following harvest of tallgrass prairie at different times throughout the growing season. Department of Agronomy-College of Agriculture. Kansas State University, Manhattan, Kansas, pp. 109.

NASS. 2004. New York turfgrass survey. NY Department of Agricolture and Marketing, Albany, New York.

Nowak D.J. 1993. Atmospheric carbon reduction by urban trees. Journal of Environmental Management 37: 207-217.

Nowak D.J., Crane D.E., Stevens J.C., Hoehn R.E., Walton J.T. and Bond J. 2008. A ground-based method of assessing urban forest structure and ecosystem services. Arboriculture and Urban Forestry 34: 347-358.

Nowak D.J., Greenfield E.J., Hoehn R.E. and Lapoint E. 2013. Carbon storage and sequestration by trees in urban and community areas of the United States. Environmental Pollution 178: 229-236.

Nowak D.J., Stevens J.C., Sisinni S.M. and Luley C.J. 2002. Effects of urban tree management and species selection on atmospheric carbon dioxide. Journal of Arboriculture 28: 113-122.

Odum E. and Barrett G. 2005. Fundamentals of ecology. Thomson Brooks/Cole, pp. 598.

Odum E. 1997. Ecology: a bridge between science and society. Sinauer Associates Incorporated, pp. 330.

Parton W.J., Hartman M., Ojima D. and Schimel D. 1998. DAYCENT and its land surface submodel: description and testing. Global and planetary Change 19: 35-48.

Parton W.J., Stewart J.W.B. and Cole C.V. 1987. Dynamics of C, N, P and S in grassland soils: a model. Biogeochemistry 5: 109-131.

Pataki D.E. 2006. Controls on C and N cycling in a southern California urban turfgrass ecosystem. Soil Carbon and California's Terrestrial Ecosystems 2005221.

Post W.M., Izaurralde R.C., Jastrow J.D., McCarl B.A., Amonette J.E., Bailey V.L., Jardine P.M., West T.O. and Zhou J. 2004. Enhancement of carbon sequestration in US soils. Bioscience 54: 895-908.

Post W.M. and Kwon K.C. 2000. Soil carbon sequestration and land-use change: processes and potential. Global change biology 6: 317-327.

Potere D. and Schneider A. 2007. A critical look at representations of urban areas in global maps. GeoJournal 69: 55-80.

Pouyat R., Groffman P., Yesilonis I. and Hernandez L. 2002. Soil carbon pools and fluxes in urban ecosystems. Environmental pollution 116: S107-S118.

Pouyat R.V., McDonnell M.J. and Pickett S.T. 1997. Litter decomposition and nitrogen mineralization in oak stands along an urban-rural land use gradient. Urban Ecosystems 1: 117-131.

Pouyat R.V., Russell-Anelli J., Yesilonis I.D. and Groffman P.M. 2003. Soil carbon in urban forest ecosystems. The potential of US forest soils to sequester carbon and mitigate the greenhouse effect. Boca Raton, FL: CRC Press, pp. 347-362.

Pouyat R.V., Yesilonis I.D. and Golubiewski N.E. 2009. A comparison of soil organic carbon stocks between residential turf grass and native soil. Urban Ecosystem 12: 45-62.

Pouyat R.V., Yesilonis I.D. and Nowak D.J. 2006. Carbon storage by urban soils in the United States. Journal of environmental quality, pp. 1566-1575.

Qian Y., Bandaranayake W., Parton W.J., Mecham B., Harivandi M.A. and Mosier A.R. 2003. Long-term effects of clipping and nitrogen management in turfgrass on soil organic carbon and nitrogen dynamics. Journal of Environmental Quality 32: 1694-1700.

Qian Y. and Follett R.F. 2002. Assessing soil carbon sequestration in turfgrass systems using long-term soil testing data. Agronomy Journal 94: 930-935.

Qian Y., Follett R.F. and Kimble J.M. 2010. Soil organic carbon input from urban turfgrasses. Soil Science Society of America Journal 74: 366-371.

Raciti S.M., Groffman P.M., Jenkins J.C., Pouyat R.V., Fahey T.J., Pickett S.T. and Cadenasso M.L. 2011. Accumulation of carbon and nitrogen in residential soils with different land-use histories. Ecosystems 14: 287-297.

R&A, 2014. Playing golf, the golf course. URL consulted on 2014 January 16. http://www.randa.org/en/Playing-Golf/Quick-Guide-to-the-Rules.aspx

Randerson J., Chapin Iii F., Harden J., Neff J. and Harmon M. 2002. Net ecosystem production: a comprehensive measure of net carbon accumulation by ecosystems. Ecological applications 12: 937-947.

Rossiter D.G. 2007. Classification of urban and industrial soils in the world reference base for soil resources. Journal of Soils and Sediments 7: 96-100.

Rowntree R.A. and Nowak D.J. 1991. Quantifying the role of urban forests in removing atmospheric carbon dioxide. Journal of Arboriculture 17: 269-275.

Roy S., Byrne J. and Pickering C. 2012. A systematic quantitative review of urban tree benefits, costs, and assessment methods across cities in different climatic zones. Urban Forestry & Urban Greening, 11: 351-363.

Schlesinger W.H. and Andrews J.A. 2000. Soil respiration and the global carbon cycle. Biogeochemistry 48: 7-20.

Schulze E.-D., Wirth C. and Heimann M. 2000. Managing forests after Kyoto. Science(Washington) 289: 2058-2059.

Schulze E.D., Beck E. and Müller-Hohenstein K. 2005. Plant ecology. Springer, pp. 702.

Selhorst A. and Lal R. 2013. Net Carbon Sequestration Potential and Emissions in Home Lawn Turfgrasses of the United States. Environmental management 51: 198-208.

Selhorst A.L. and Lal R. 2011. Carbon budgeting in golf course soils of Central Ohio. Urban Ecosystems 14: 771-781.

Singh M.H. 2007. Soil Organic Carbon pools in turfgrass systems in Ohio. The Ohio State University, pp. 160.

Stehfest E., Heistermann M., Priess J.A., Ojima D.S. and Alcamo J. 2007. Simulation of global crop production with the ecosystem model DayCent. Ecological Modelling 209: 203-219.

Stewart C.E., Paustian K., Conant R.T., Plante A.F. and Six J. 2007. Soil carbon saturation: concept, evidence and evaluation. Biogeochemistry 86: 19-31.

Takahashi T., Amano Y., Kuchimura K. and Kobayashi T. 2008. Carbon content of soil in urban parks in Tokyo, Japan. Landscape and Ecological Engineering 4: 139-142.

Tansley A.G. 1935. The use and abuse of vegetational concepts and terms. Ecology 16: 284-307.

The American Heritage, 2013. s.v. turf, Dictionary of the English Language. URL consulted on 2013 October 21. http://www.ahdictionary.com/word/search.html?q=turf

Townsend-Small A. and Czimczik C.I. 2010. Carbon sequestration and greenhouse gas emissions in urban turf. Geophysical Research Letters 37, pp.5.

Turgeon A. 1985. Turfgrass management. Reston Publishing Company, pp. 396.

Vinlove F.K. and Torla R.F. 1994. Comparative estimations of US home lawn area. Journal of Turfgrass Management 1: 83-97.

Wagai R., Brye K.R., Gower S.T., Norman J.M. and Bundy L.G. 1998. Land use and environmental factors influencing soil surface CO_2 flux and microbial biomass in natural and managed ecosystems in southern Wisconsin. Soil Biology and Biochemistry 30: 1501-1509.

Wheeler K. and Nauright J. 2006. A global perspective on the environmental impact of golf. Sport in Society 9: 427-443.

Wolf S. 2010. Carbon dioxide and water vapour fluxes of tropical pasture and afforestation: seasonal variations of net ecosystem exchange and carbon sequestration potentials. Eth Zurich, pp. 163.

Wood H. 2010. Considering Carbon Emissions and Sequestration in Turf Grass and Amenity Management. Landscape & Environmental Services Ltd. London.

Wu J. and Bauer M.E. 2012. Estimating Net Primary Production of Turfgrass in an Urban-Suburban Landscape with QuickBird Imagery. Remote Sensing 4: 849-866.

Zhang Y. 2012. Simulated carbon and nitrogen dynamics in turfgrass systems using the DAYCENT model-Chapter2: Simulation of N_2O emissions and estimation of global warming potential in turfgrasses using the DAYCENT model. Department of Horticulture and Landscape Architecture. Colorado State University, Fort Collins, Colorado, pp. 78.

Zhang Y., Qian Y., Bremer D.J. and Kaye J.P. 2013. Simulation of Nitrous Oxide Emissions and Estimation of Global Warming Potential in Turfgrass Systems Using the DAYCENT Model. Journal of Environmental Quality.

Zhou X., Wang X., Tong L., Zhang H., Lu F., Zheng F., Hou P., Song W. and Ouyang Z. 2012. Soil warming effect on net ecosystem exchange of carbon dioxide during the transition from winter carbon source to spring carbon sink in a temperate urban lawn. Journal of Environmental Sciences 24: 2104-2112.

CHAPTER 3
Estimated net ecosystem exchange (NEE) of turfgrass at different management intensities in a golf course in the province of Verona

3.1 Introduction

Because of the increasing greenhouse gases (GHGs) concentration in the atmosphere and the related global warming (IPCC, 2013), it is crucial to understand the mechanisms driving carbon (C) sequestration in terrestrial ecosystems in order to study their response to the changing environment and to minimize the C footprint of agriculture and urban green areas.

In natural and agricultural systems, atmospheric CO_2 is fixed by photosynthesis. A portion of C fixed by photosynthesis is transferred to soil as root exudates or plant litter. By increasing plant productivity and belowground carbon allocation we can enhance CO_2 removal from the atmosphere, allowing long term C sequestration within the soil. In managed ecosystems this can be coupled with the adoption of more sustainable farming practices aimed at reducing GHGs emissions and preserving soil organic carbon (SOC) (Lal, 2004).

Urban vegetation, although not usually managed to achieve maximum productivity, follows the same mechanisms of C fixation and transfer to the soil. Appropriate planning, design and maintenance practices are needed to ensure the optimal conditions for vegetation to perform its functions (Bartlett and James, 2011) and increase ecosystem biomass and related flows of C to the soil.

Turfgrass plays an important role in urban vegetation, due to its extension and ubiquity (See Section 2.2.3).

Understanding the role of turfgrasses and its management on C cycle is important in order to know the real benefits of the whole urban forests towards the climate change challenge.

Studies on C sequestration of urban trees have been increasingly published since the 1990's (McPherson, et al., 1997; Nowak, et al., 2002; Nowak, et al., 2008; Russo, et al., 2014).

In the last decade, several studies investigated the C sequestration potential of turfgrass ecosystems as well (Qian and Follett, 2002; Bandaranayake, et al., 2003; Huyler, et al., 2013; Selhorst and Lal, 2013), applying different approaches and

methodologies: SOC chrono-sequences, isotopes techniques, remote sensing-, ecosystemic-, LCA-models, direct measurements of gas exchange (an extended review of these studies has been presented in Chapter 2). However, the role of turfgrass in Continental and Mediterranean climates is not yet clearly understood.

The aim of the present study is therefore to improve the understanding of the mechanisms underlying C fluxes in a turfgrass ecosystem and to assess its C sequestration potential by estimating the annual C budget.

3.1.1 The role of golf courses on urban landscape

In the two last decades environmental benefits of urban green spaces (Laghai and Bahmanpour, 2012) and turfgrasses (Beard and Green, 1994) have been widely studied (Table 3.1) and are summarized in Table 3.2. Turfgrasses are extensively used in urban and suburban landscapes, including residential gardens, parks, commercial and institutional lawns, recreational facilities, green belts, sports fields and golf courses.

A golf course is "*a large area of grassed land with a series of nine or eighteen holes, used for playing golf*" (The Cambridge Dictionary Online, 2013).

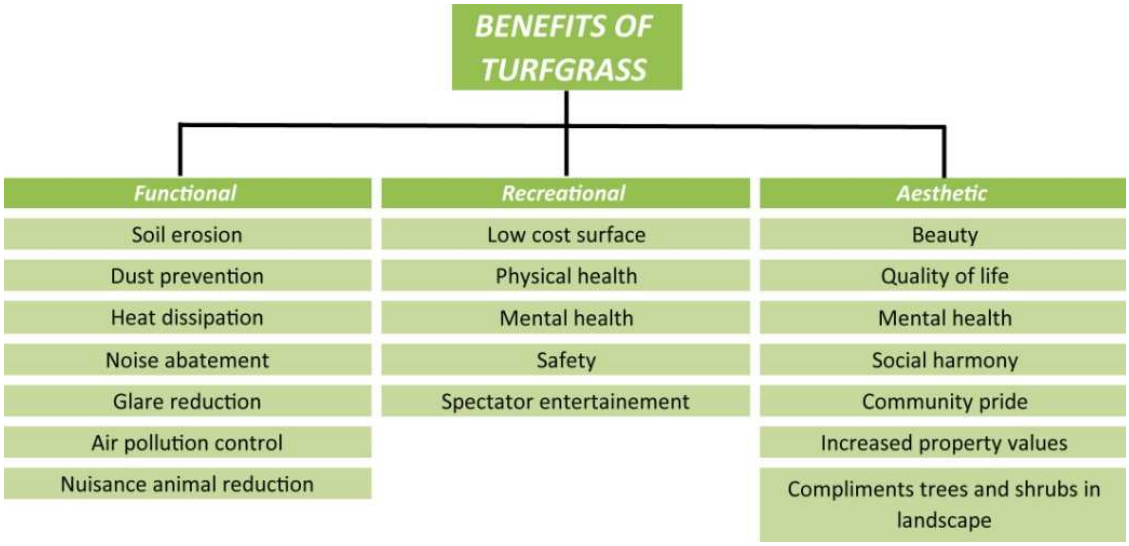
The presence of golf courses in urban landscapes is increasing globally, becoming more and more important both environmentally and economically.

Several studies (Tanner and Gange, 2005; Hammond and Hudson, 2007; Sorace and Visentin, 2007; Colding and Folke, 2009) reported the increase in the biodiversity of plants, insects, micro fauna and birdlife on golf courses. These ecosystems, even if situated close to urban conglomerates, are part of the rural framework and may play a central role in biodiversity preservation. However, in this context, turfgrasses are viewed as intensive agriculture lands, due to fertilizers and pesticides leaching, high CO_2 emissions from maintenance activities, etc. In the last years several golf courses have adopted environmental friendly guidelines in order to minimize possible detrimental environmental consequences. Some of the strategies suggested for a more sustainable management of turfgrasses are: 1) establishment of warm-season species, when possible; 2) adoption of appropriate techniques of maintenance; 3) reduction of areas with "low playing components", characterized by intensive management; 4) increase of the wild areas (uncultivated areas) in order to create suitable conditions for the preservation and survival of the wild fauna.

Table 3.1: Urban green space functions and services. Reworked graphic version from Laghai and Bahmanpour (2012).

Function / Service	*Source*
Increase physical activity prevalence	Van Sluijs et al., 2007
Outdoor recreation	Busiey & Coles, 1995; Roovers et al., 2002
Healthy	UN-World Health Organization, 1993
Social services	Escobedo et al., 2008
Economic services	Conway & Urbani, 2007; McPherson et al., 2005
Effective interceptors and reflectors of isolation (short wave solar radiation)	Chen & Jim, 2008
Improving thermal energy	Jeusen et al., 2003; Mcpherson et al., 1997
Lower indoor air temperature	Mcpherson et al., 2006
Heating saving	Mcpherson et al., 2006; Heisler, 1986
Reduce air pollution	Chen & Jim, 2008
Alleviate poor air quality by absorbing a gaseous pollutants (e.g., O3, NO2), Intercepting particulate matter (e.g., PM10 such as dust, ash, pollen), Releasing oxygen, Reducing the frequency of conditions leading Moderating local air temperatures	McPherson et al., 2006; Nowak et al., 2006
Reduce the risk of heat-related illnesses	Blum et al., 1998
Reduce stress	Ulrich, 1981; Woo et al., 2009
Reduce in the amount of storm water runoff	Xiao & McPherson, 2002
Heating & air conditioning	McPherson et al., 2006
Intercept and store rainfall, reducing runoff volumes and delaying the onset of peak flows, Reduce flooding, hazards, surface pollutant wash out and pollutant loading of rivers and lakes	McPherson et al., 2006
Increase the attractiveness of communities	Chen & Jim, 2008
Reduce noise	McPherson & Simpson, 2002
Improve wildlife habitat Provide recreational opportunities	Nowak & Dwyer, 2002
Air and water purification; wind and noise filtering Micro climate stabilization; social and psychological services	Ulrich, 1981
Increase neighborhood desirability	Pepper et al., 2007
Providing recreational opportunities Enhancing aesthetic values	Nowak & McPherson, 1993
Support biodiversity	Gaston et al., 2005; Smith et al., 2005
Provide a sense of peace and tranquility	Keplan, 1985; Song et al., 2007
Increase the sale price of nearby homes	Conway et al., 2008; Chen & Jim, 2006

Table 3.2: Summary of benefits derived from turf. Reworked graphic version from Beard and Green (1994).



In 2005 the number of golf courses in the world exceeded 31,500 (Tanner and Gange, 2005). In2008 there were more than 35,100 golf courses globally (Saito, 2010) (Table 3.3), with the USA accounting for 50% of the total, and the top five countries (USA, UK, Japan, Canada and Australia) accounting for 76%. Assuming a typical golf course area of 60–80 ha, the spatial scale can be estimated to be 21,060-28,080 km^2 worldwide.

In 2011 in Europe there were 6,740 regular golf courses, with 4,389,740 affiliated players (KMPG, 2011). The UK and Ireland account for 44%, and the top five countries (UK, Ireland, Germany, France and Sweden) account for 71% (Figure 3.1). In 2012 in Italy the number of golf courses was 410 (Federazione Italiana Golf, 2013): 66% was situated in the North, 25% in the Centre and 9% in the South and islands (Figure 3.2). Lombardia was the region with the highest number of golf courses (68), while Basilicata is the last region of the list with just 1 golf course. The whole area of the Italian golf courses range between 24.6 and 32.8 km^2.

Table 3.3: Number of golf courses and distribution in the world in 2008. Reworked graphic version from Saito (2008).

Country	Number of Golf courses	Percentage
USA	17,672	50%
UK	2,752	8%
Japan	2,442	7%
Canada	2,300	7%
Australia	1,500	4%
Germany	684	2%
France	559	2%
China	500	1%
Sweden	480	1%
South Africa	450	1%
Rest of the world	5,773	17%
Total	*35,112*	*100%*

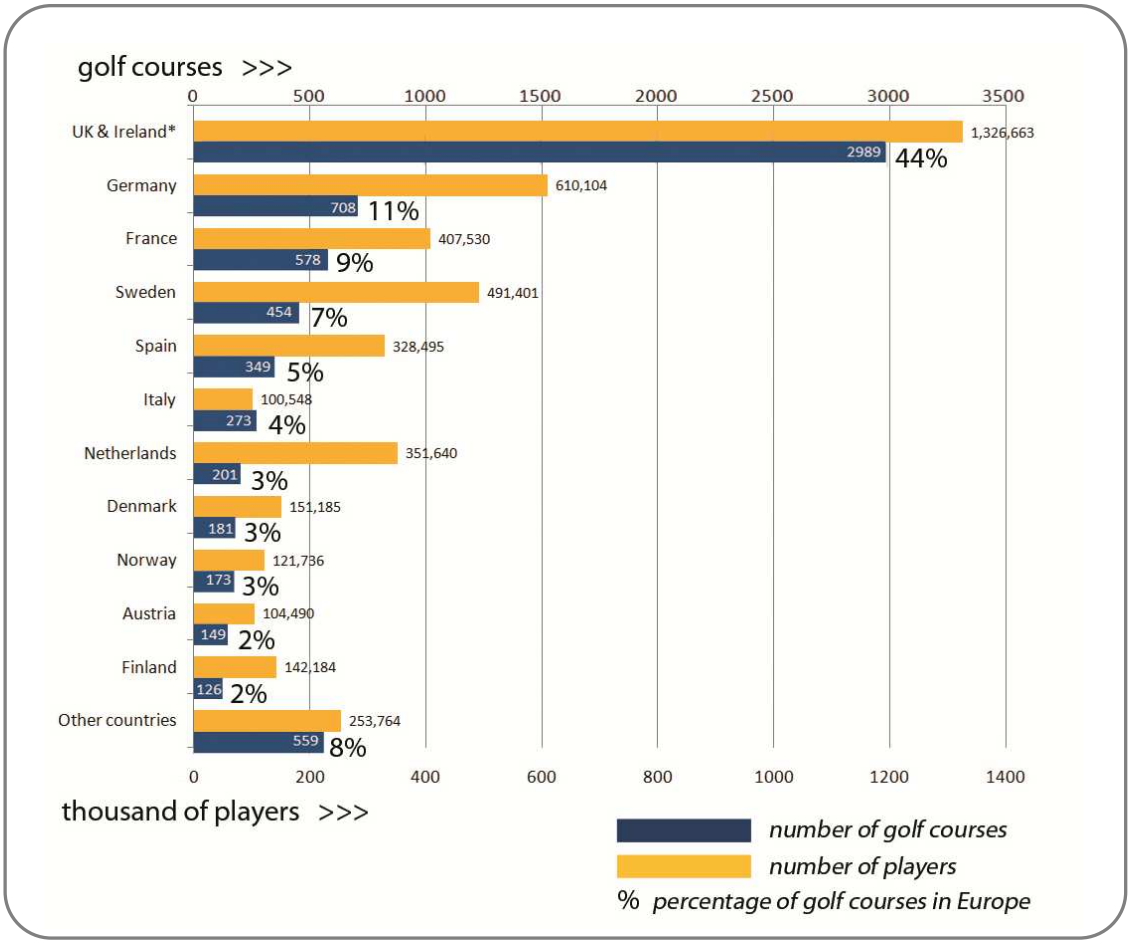


Figure 3.1: Number of golf courses and distribution in Europe in 2011. Reworked graphic version from KPGM (2011). *Please note that, as opposed to most European countries, in the UK & Ireland golfers do not have to register to play. The unregistered golfers in these statistics are not considered.

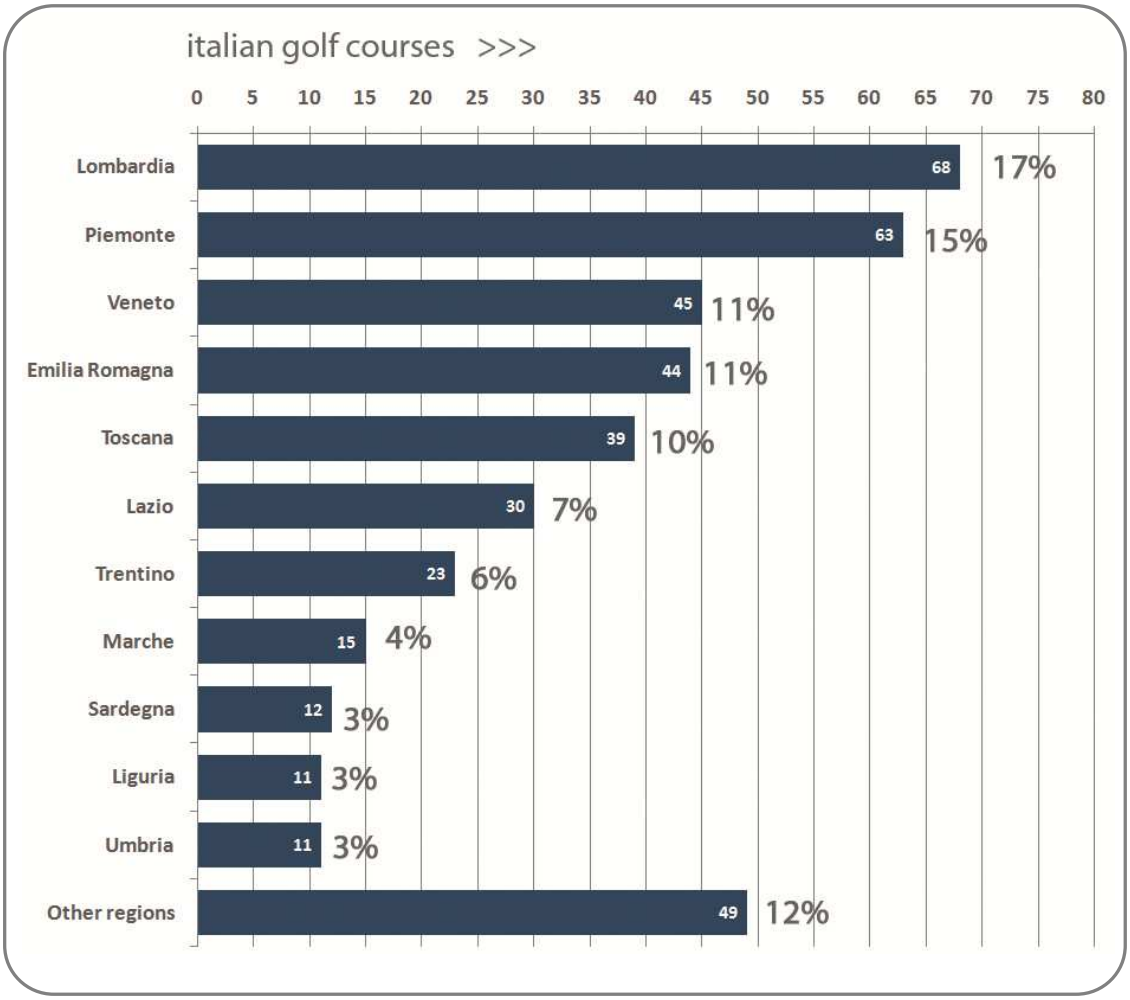


Figure 3.2: Number of golf courses, distribution and percentage in Italy in 2012, elaborated from Federazione Italiana Golf (2013).

3.1.2 Carbon fluxes and productivity of ecosystems: theories and definitions

In ecological terms the word "photosynthesis" is used to denote the C fixed by gross photosynthesis minus the C lost by photorespiration. Some of that photosynthetically fixed C is lost by internal plant metabolism, through autotrophic respiration (R_a). Heterotrophic respiration (R_h) refers to the C respired by heterotrophic organisms (soil microorganism, animals etc.). Soil CO_2 efflux is thus comprised of R_a and R_h and the quantification of their relative contribution, which varies seasonally among ecosystems and latitudes is methodologically challenging to achieve (Kirschbaum, et al., 2001) (Figure 3.3).

The exchange of CO_2 between ecosystem and the atmosphere reflects the balance between photosynthesis (by plants) and respiration (R_a and R_h) (Aber and Melillo, 2001). This balance, called net ecosystem production (NEP) has become a critical characteristic of terrestrial ecosystems (Figure 3.4) in order to study their response to the increase in atmospheric CO_2 concentration.

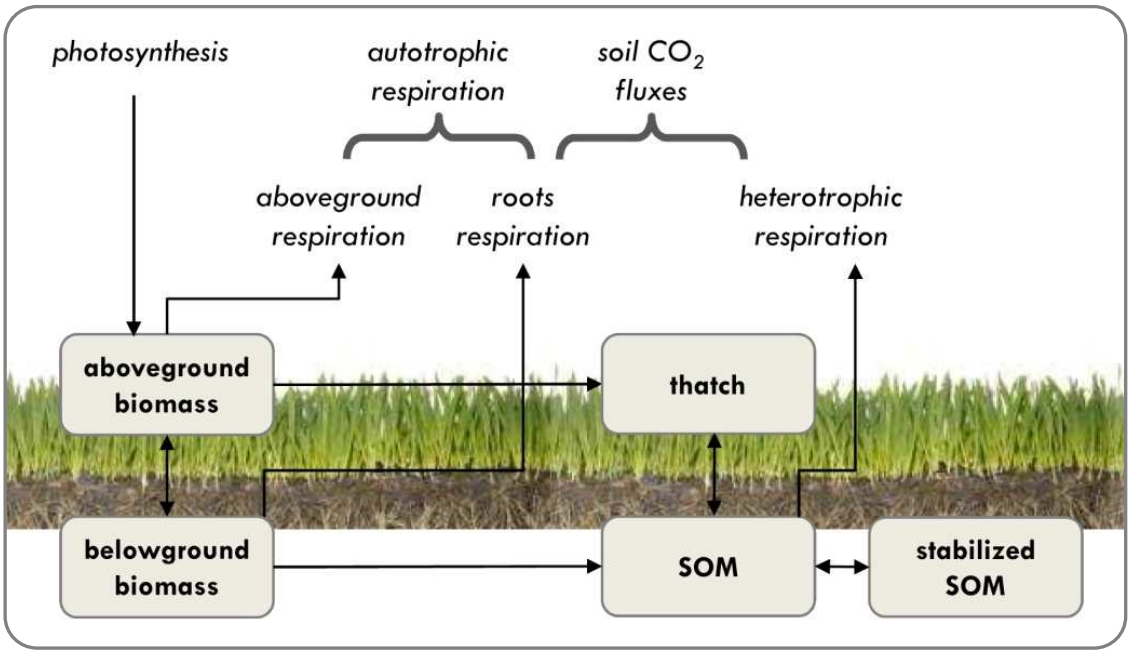


Figure 3.3: Major fluxes (arrows) and pools (boxes) of turfgrass ecosystem. Modified from Kirschbaum and colleagues (2001).

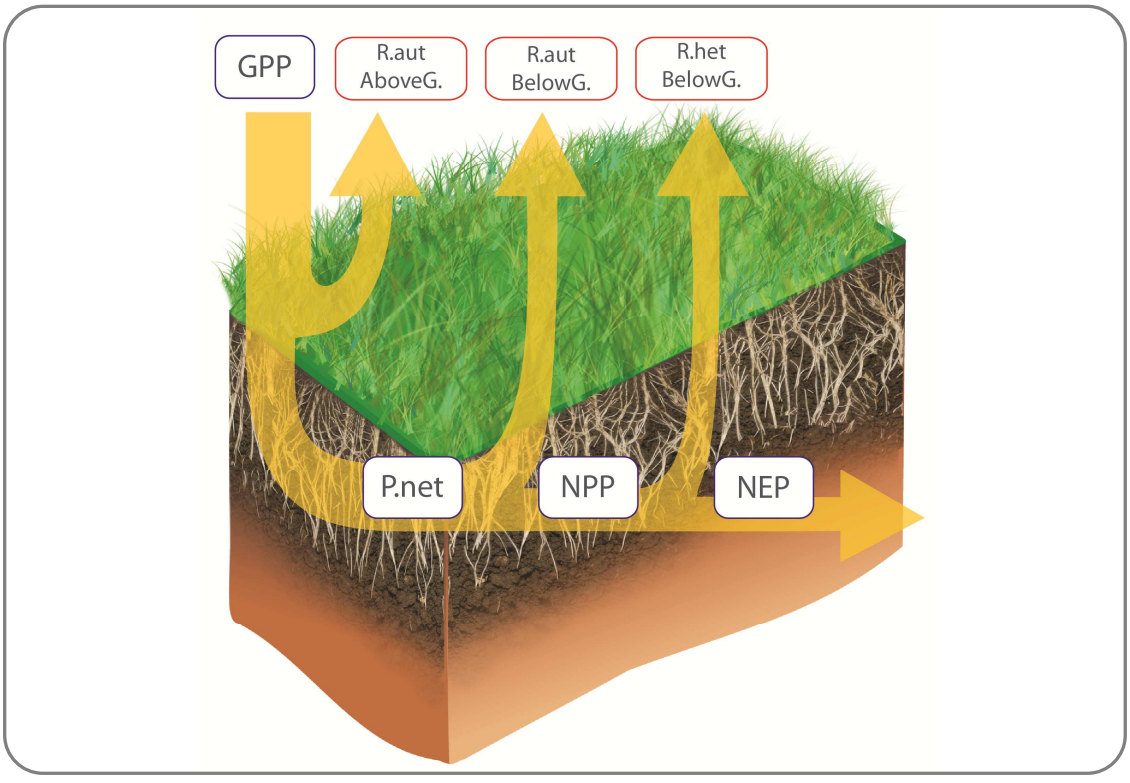


Figure 3.4: Schematic representation of turfgrass C uptake. GPP=gross primary productivity; R.autAboveG.=aboveground autotrophic respiration; R.autBelowG.=below ground autotrophic respiration; R.het BelowG.=belowground heterotrophic respiration; P.net=net photosynthesis; NPP=net primary productivity; NEP =net ecosystem productivity. Modified from Schulze et al., 2005.

Another fundamental property of an ecosystem is net primary productivity (NPP), which refers to the net production of organic carbon (OC) by plants and it is usually measured over a period of one year, or more (Kirschbaum, et al., 2001). The term includes not only the growth of primary producers, but also the C transfer to roots symbionts (mycorrhizal fungi for example), the production of root exudates and plants volatile organic compounds (VOCs). Net primary productivity is equivalent to GPP minus autotrophic respiration (R_a):

$$NPP = GPP - R_a \quad \textbf{(eq. 3.1)}$$

Net ecosystem productivity, or net ecosystem exchange (NEE) with opposite sign, is defined as the difference between the amount of C fixed by photosynthesis (GPP) and the total ecosystem respiration ($R_{eco}=R_a + R_h$):

$$NEP=-NEE= GPP - (R_a + R_h) \quad \textbf{(eq. 3.2)}$$

Net ecosystem productivity has been also defined by some authors as the net rate of C accumulation in ecosystems (Woodwell and Whittaker, 1968). The two definitions are equal in a simple ecosystem model in which the rate of C accumulation results from the balance between photosynthesis and respiration. This is a reasonable approximation in long-term balances for natural ecosystem studies that do not consider other terms. In other ecosystems, such as cities and streams, lateral flows of C and energy can be the major determinants of net ecosystem C balance (Chapin III, et al., 2006). Therefore, Lovett and colleagues (2006) stated that the net C accumulation rate in ecosystems does not equal the imbalance between GPP and R_{eco}. Net ecosystem productivity might thus not be confused with the rate of C accumulation (Figure 3.5), which, expanding equation 3.2 can be written as:

$$NEP = GPP - (R_a + R_h) \quad \text{and}$$

$$GPP - (R_a + R_h) = \Delta C_{org} - Imp + Exp + Ox_{nb} \quad \text{then}$$

$$NEP = \Delta C_{org} - Imp + Exp + Ox_{nb} \quad \text{then}$$

$$NEP + Imp = \Delta C_{org} + Exp + Ox_{nb} \quad \text{then}$$

$$\Delta C_{org} = GPP + Imp - Exp - (R_a + R_h) - Ox_{nb} \quad \textbf{(eq. 3.3)}$$

Table 3.4: Components of the equations 3.1, 3.2 and 3.3.

GPP	Gross Primary Production
ΔC_{org}	Change in OC Storage in the Ecosystem
Imp and Exp	Import and Export of OC
R_a and R_h	Autotrophic and Heterotrophic Respiration
Ox_{nb}	Nonbiological Oxidation of C

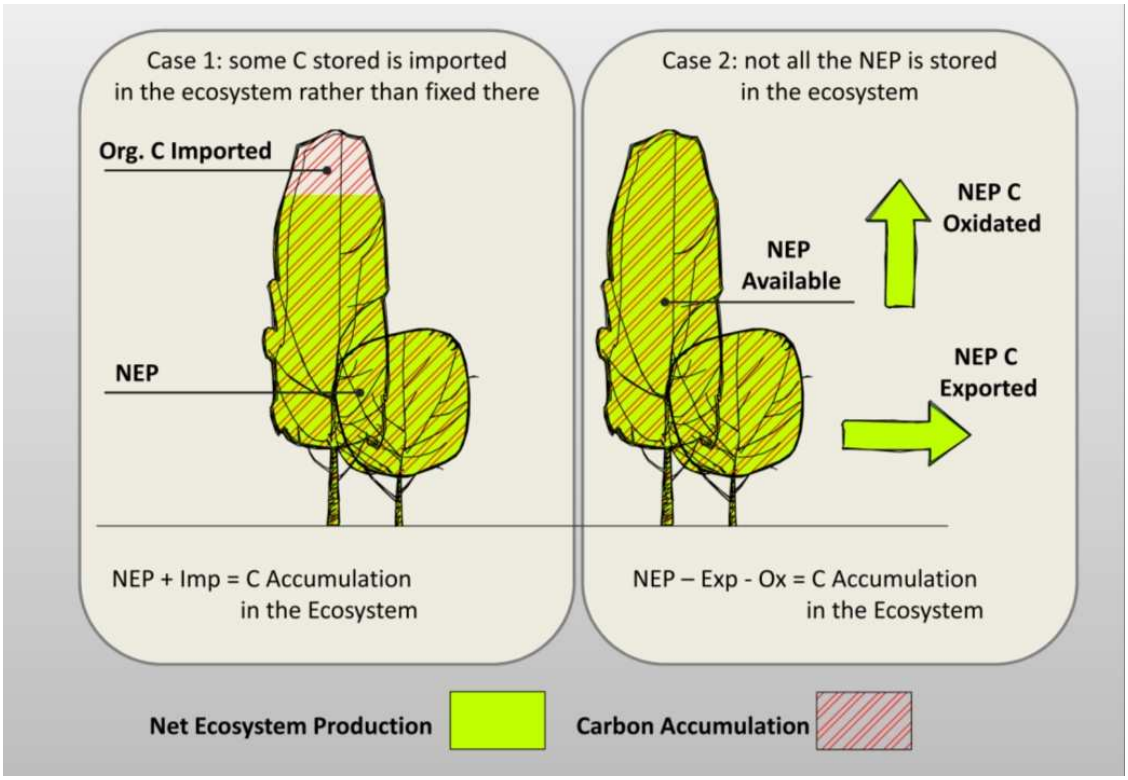


Figure 3.5: The rate of C accumulation is the same in both cases: In the figure on the left is formed by NEP + Imp; in the figure on the right is formed by NEP-Exp-Ox.

The C accumulation rate is defined as "*the change in the C pool per unit of time*"(Chapin III, et al., 2006), while NEP is "*the amount of OC fixed in an ecosystem that is not respired there and it is therefore available for accumulation, export, non-biological oxidation*"(Chapin III, et al., 2006). Net ecosystem productivity may be a good approximation of the C accumulation rate within the system if inputs and outputs of OC other than GPP and R_{eco} are negligible (Figure 3.5). Many studies have used NEP and OC accumulation interchangeably. However, in many cases, inputs and outputs terms cannot be ignored.

Regardless Chapin (2006) proposes a new term in order to understand the net rate of C accumulation: net ecosystem carbon balance (NECB). Net ecosystem carbon balance represents the whole ecosystem C balance from physical, biological and

anthropogenic sinks and source. It considers C input and output from the ecosystem over a specified time interval.

$$NECB=\Delta C/\Delta t \quad \textbf{(eq. 3.4)}$$

Net Ecosystem Exchange is defined by atmospheric scientist as a C input to the atmosphere, or as the CO_2 flux from the ecosystem to the atmosphere. Values are negatives when the system sequesters C (photosynthesis) and positives when the system releases C (respiration). On the contrary, NEP and NECB are defined by ecologists as C input to ecosystems, and have opposite sign than NEE.

The present study does not consider the entire biome over large temporal and spatial scales but some representative areas sampled several times over one year to account for seasonal variability and in different positions to account for spatial heterogeneity. We assumed that in our turfgrass ecosystem non-biological oxidation, import and export of C are negligible if compared to the magnitude of other fluxes. We therefore considered NEE to be a good approximation of the C accumulation rate within the system compared to NECB and ΔC_{org}, which are not adequate to describe flux exchanges of our study.

3.1.3 Carbon fluxes and productivity of turfgrass: mechanisms and underlying factors

Gross primary production (GPP) and ecosystem respiration (R_{eco}) are the most important C fluxes in turfgrass ecosystem as in many other terrestrial ecosystems. During the day, in light conditions, GPP usually exceeds R_{eco}, and NEP results positive (eq. 3.2). On the contrary, during the night, in dark conditions, R_{eco} is the only active flux, thus NEP results negative.

Photosynthesis, respiration and decomposition change during the day and the seasons, depending on different factors (Figure 3.6):

- availability of soil resources (water and nutrients),
- external disturbance (human activities),
- leaf area index (LAI),
-length of the photosynthetic season and climate conditions (temperature and light).

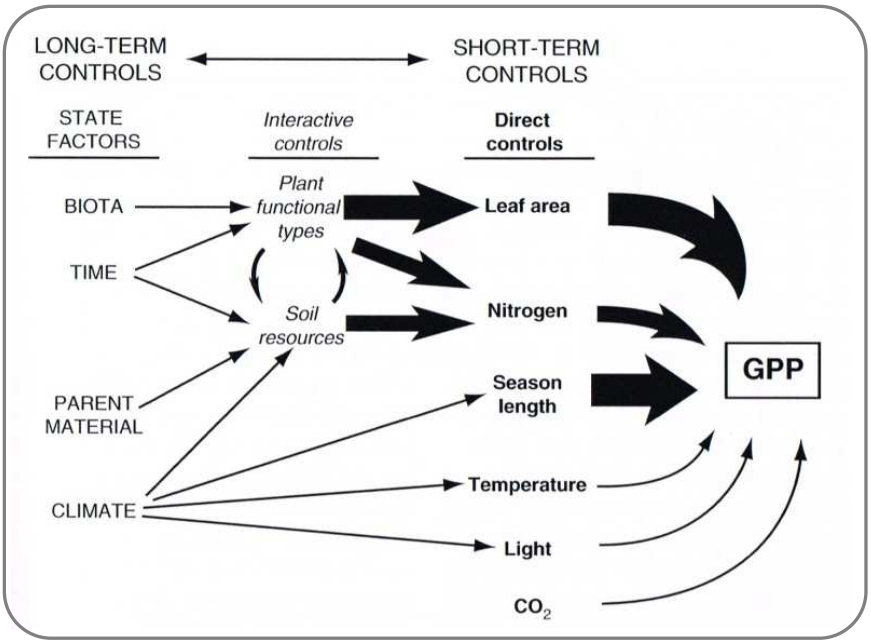


Figure 3.6: The major factors governing temporal and spatial variation in GPP in ecosystems (Chapin III and Matson, 2011).

The main environmental factors affecting rate of photosynthesis in turfgrass are light intensity, temperature and CO_2 concentration (Chapin III and Matson, 2011).

The light influence directly the photosynthesis activity: as light intensity increases, the rate of photosynthesis increases proportionately, until it reaches a saturation value above which, further increases in light intensity, do not cause rises in photosynthetic rate. Cool-season grass metabolism (C3) includes also photorespiration, which is active exclusively during the daylight, like photosynthesis, whereas respiration activity runs 24 hours long. At low light intensities, respiration can be greater than photosynthesis, therefore the system produce less organic matter (OM) than the respiration consume (Venturelli and Virli, 1995).

The relationships between NEE and photosynthetic active radiation (PAR) in turfgrass (Zhou, et al., 2012) or photosynthetic photon flux density (PPFD) in grassland (Flanagan, et al., 2002), have been previously well described by rectangular hyperbolic functions.Temperature influences both photosynthesis and respiration. At low temperatures, photosynthesis is limited because chemical reactions, catalyzed by enzymes, go slowly. As the enzymes approach their optimal temperatures, the rate of enzymatic activity increases and so does the photosynthesis. Above the optimum for temperature the rate decreases and begins to limit photosynthesis until at high temperatures photorespiration becomes the limiting factor. Further extreme conditions cause enzyme inactivation and destruction of pigments (Chapin III and Matson, 2011).

Photosynthesis and respiration of plants react in different way to the change of temperatures. Autotrophic respiration is more sensitive and increases at a greater rate than photosynthesis, with consequences on the overall CO_2 emissions from the ecosystem. Furthermore extended periods of drought reduce NEP. On the other hand, climate warming permits at ecosystem photosynthesis to start early, increasing C uptake, and to accelerate the winter limited metabolism of certain species of turf.

Generally, exponential models are used to describe the relation between temperature and respiration (Fang and Moncrieff, 2001). The Q10 value, which defines the temperature dependence or sensitivity to temperature variation of respiration, is the factor by which respiration is multiplied when temperature increases by 10°C. The Q10 is then the ratio between respiration at a reference temperature (T ref.) and the respiration at a temperature 10°C lower (T -10°C). Q10 of a single, membrane bound, plant respiratory enzyme is usually comprised between 1.5 and 2.0 (Davidson, et al., 2006), while in terrestrial ecosystem Q10 of soil respiration ranged from 1.3 to 3.3 (Raich and Schlesinger, 1992). The apparent Q10 of ecosystem and soil respiration is the results of several temperature-sensitive processes, including enzyme activity, diffusion of O_2 and soluble carbon substrates through soil air and water and across cellular membranes, and growth of microbial populations and root tissues, that can have multiplicative effects increasing or decreasing the values of Q10 (Davidson et al. 2006).

A few studies tested how the expected increase in soil temperature could affect NEE on turfgrass (Pataki, 2006; Zhou, et al., 2012). Zhou (2012), simulated global warming effect on urban lawns (*Zoysia japonica*) from winter to springtime. The increase of soil temperature (5°C) influenced the turfgrass, transforming the system from a source to a sink of C. During the transition period from winter to spring, photosynthesis feedback was more sensitive than respiration to the increase of temperature. NEE estimates in this period pinpointed the switch from winter C source to spring C sink. The increase of temperature (3.5°C) on plots of *Festuca arundinacea* in California (Pataki, 2006), showed an increase in rates of ecosystem respiration, particularly in winter when heterotrophic respiration was dominant, and a significant decrease in NEE.

Frequently cultural operations carried out on turfgrass, especially mowing, characterize turf physiology compared to similar herbaceous vegetation. According to Turgeon (1985) mowing is detrimental to turfgrass physiology because "*it causes a temporary cessation of root growth, reduces carbohydrate production and storage, creates ports of entry for disease-causing organisms, temporarily increases water loss from cut leaf ends, and reduces water absorption by the roots*". However, the species used for turfgrasses evolved under conditions of periodic partial defoliation (animals and fires) and have adapted their physiology and

morphology for responding to these external disturbances. Therefore mowing has not a unilateral impact (positive or negative) on turfgrass, but should be planned and adapted to environmental conditions to obtain maximum quality, serving an aesthetic and utilitarian function.

Mowing frequency is regulated by physiological aspects: as the loss of too much photosynthetic surface would cause adverse physiological responses (Hull, 2000), it is recommended to do not remove more than one third of the aboveground tissues in a single operation (Turgeon, 1985). Despite mowing at adequate height increase tiller density (Razmjoo, et al., 1996), the removal of leaf area reduces the plant's ability to capture PAR and carbohydrates reserves are mobilized to support re-growth, diminishing the tolerance to stress conditions. However, Hull (2000) did not find significant effect of mowing frequency on turfgrass productivity.

Turfgrass subjected to a continuous mowing achieve a dynamic equilibrium between roots and shoots and does not suffer particular stress. Hull (1987) found that adequate mowing can increase CO_2 fixed from the remaining leaves by 10% as shown in Figure 3.7, suggesting that photosynthesis compensates for partial defoliation due to the lower leaf shading and to the greater demand by leaves for photosynthate from non-green plant parts (root and crown tissue).

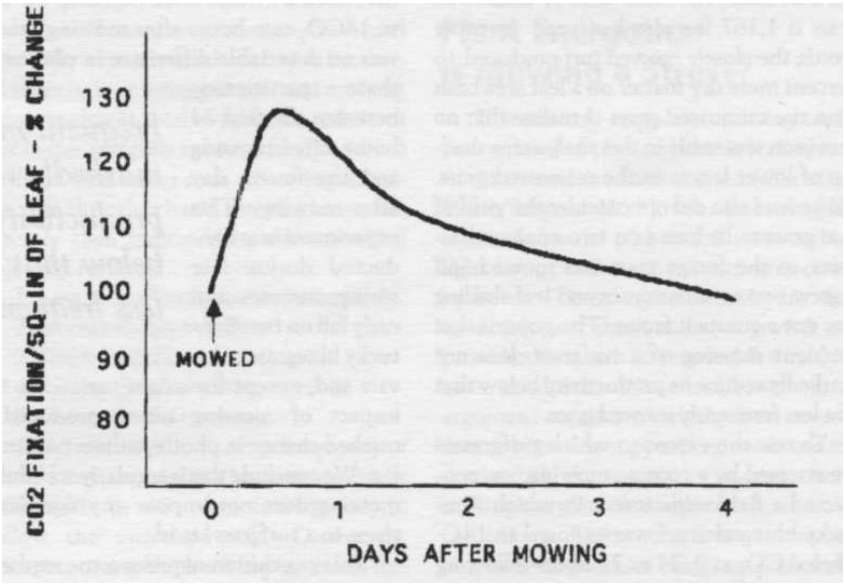


Figure 3.7: Change in net photosynthetic rate of turf per unit leaf area in response to mowing (Hull, 1987).

3.1.4 Methodologies for gas exchange estimates

Correct measurements of CO_2 fluxes between vegetation and the atmosphere are a prerequisite to calculate the C balance (Kutzbach, et al., 2007). The methodological approaches adopted for the calculation of the ecosystem C balance are two: the whole-system balance (micrometeorological methods) and the small-chamber enclosures (SC) (Baldocchi, et al., 1988).

The micrometeorological approaches include all those methods that measure only the net CO_2 exchange between the ecosystem and the atmosphere, without quantify the single specific fluxes and pools within the system. These methods have the advantage of not creating major disturbances on the environment surrounding the plant-soil system. At the same time they are not suited for plot experiments because of their large spatial-scale. The most important methods that follow this approach are 1) the giant cylinder; 2) the aerodynamic analysis of the boundary layer and 3) the eddy covariance (EC) method.

The SC (or canopy-chamber methods) provides the measurements of the most important C fluxes within the system individually. These approaches are suitable for NEE estimates at experimental plot scale or when comparing different land use on small areas close each other.

These methods require static (closed) or dynamic (open) enclosures in which the trace gas concentration is monitored over the time with an infrared gas analyser (IRGA) and variation (increase or decrease) of CO_2concentration over the time due to plant-soil activities are used to calculate CO_2 fluxes. Photosynthesis, vegetation respiration by leaves and non-leaf tissue, roots and soil respiration are considered.

When measuring ecosystems C fluxes, the SC approach allows testing the accuracy of the micrometeorological methods by measuring the most important fluxes individually and by identifying the consistence of non-measurable pools by subtraction from the GPP (Aber and Melillo, 2001).

This approach includes two distinct methods: 1) steady-state open system and 2) transient-state closed system. The first uses open-top chambers and can monitor the plant throughout the season, with continuous measurements. Differences in CO_2 concentrations between the gas entering the chamber and that leaving the chamber are multiplied by flow rate through the chamber to obtain rates of gas exchange (Hunt, 2003). The best advantages of this system are: 1) the easier control of environmental conditions; 2) possibility to take measurements in continuous; 3) possibility to take measurements in different stages of development thanks to the non-invasive approach (Hunt, 2003).

The second method uses closed chambers that are placed over turfgrass for a very short time, for limiting disturbance on plant condition and is detailed in Section 3.1.5.

The best advantages demonstrated by closed-chamber method are: 1) relatively low costs and power consumptions (Kutzbach, et al., 2007); 2) facility of moving from place to place thanks to the portability of the instruments; 3) possibility of a series of subsequent measurements, providing replicates; 4) possibility to carry out measurement sampling by a single operator. Unlike automated chamber, human intervention is required during night-time measurements. On the other side, automated chamber cannot stand on the experimental fields if the surveys are carried out on sportive fields or golf courses, where daily maintenance operations occur.

3.1.5 Gas exchange measurement on turfgrass: small-chamber enclosure approach

Several studies utilized SC approach to determine CO_2 exchange fluxes between atmosphere and bare soil or low-stature canopies, such as tundra, grassland, forest understory vegetation, various crops and turfgrass. However, as described in details in Chapter 2, only few studies provide seasonal values of NEE on turfgrass (Allaire, et al., 2008; Livesley, et al., 2010; Zhou, et al., 2012), while none of them provide an annual estimate of NEE. We do not know European studies of NEE using SC approach on turfgrasses.

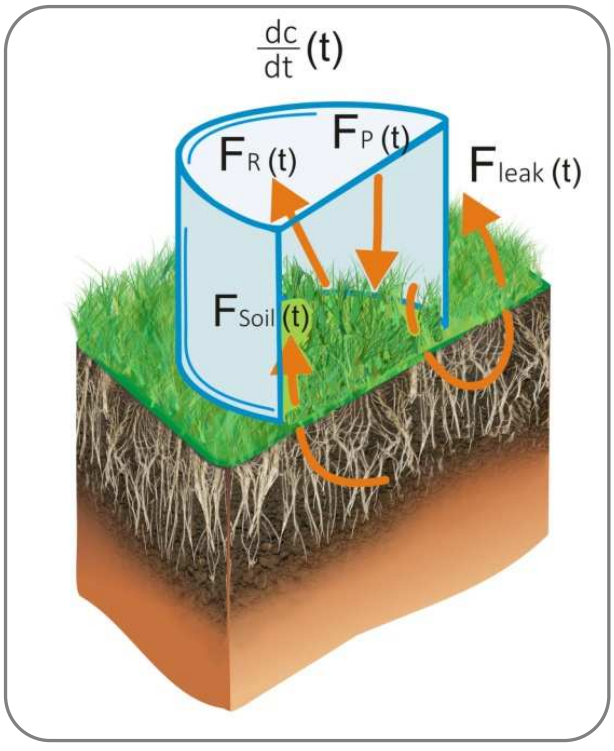


Figure 3.8: Schematic of the CO_2 fluxes in the chamber headspace which make up to the NEE, modified from Kutzbach, et al. (2007).

The measuring system (IRGA equipped with chamber) provide at each measure the CO_2 concentration change over time in the chamber headspace, that is the result of plant and soil individual processes with partly opposing directions, as illustrated in Figure 3.8. Inside the headspace, isolated from the surrounding atmosphere by transparent walls, turf photosynthesises and respires. The sum of all CO_2 fluxes in or out of the headspace represents the NEE and can be written (Kutzbach, et al., 2007) as:

$$J_c=NEE=F_{Soil}(t)+F_P(t)+F_R(t)+F_{Leak}(t) \quad \textbf{(eq. 3.5)}$$

where F_{Soil} (t) is the CO_2 efflux from the soil which originates from the respiration of soil microbes, soil animals and belowground biomass of plants, i.e. roots and rhizomes, F_P (t) is the CO_2 flux associated with the GPP of the plants, F_R (t) is the CO_2 flux associated with the dark respiration of the aboveground biomass, and F_{Leak} (t) is the CO_2 flux related to leakage directly at the chamber components or via the soil pore space.

The SC approach presents several advantages for the measurement of gas exchange of turfgrass. However, it is necessary consider also the limitations of this method, highlighted by several studies (Bremer and Ham, 2005; Livingston, et al., 2006; Kutzbach, et al., 2007; Müller, et al., 2009).

Covering vegetation with a closed chamber modify the microclimate conditions in the chamber headspace. In a little time, natural process controlling fluxes are disturbed by the changed conditions, and the CO_2 concentration gradients between soil, vegetation and air are altered. Moreover, leaks from the unsealed between the stainless steel ring and the soil surface may affect the fluxes measurement.

The NEE of turfgrass should be determined by measuring CO_2 fluxes with the chamber at neutral pressure. During the measurement, a small pump internal to the IRGA continuously circulates the air between the chamber and the gas analyzer. The suction generated by the air pumping may slightly reduce the atmospheric pressure inside the chamber. Soil respiration is sensitive to chamber pressure (Bremer and Ham, 2005). Hence, the pressure inside the chamber should be equal to the pressure outside. Despite maintaining neutral chamber pressures still appears problematic (Bremer and Ham, 2005), in the flux calculation of an automated closed-system canopy-chamber, errors that disturb the natural soil C efflux were found negligible (Steduto, et al., 2002).

The presence of leakage always exists whenever there is a gradient in gas concentration between inside and outside the chamber. Being the chamber operated as a closed-system, leaks in CO_2 represent errors in the corresponding flux rates. The lateral gas exchange established across the soil is supposed to be in equilibrium, so that CO_2 fluxes input are equal to output.

The SC method can modify the turbulence regime of the atmosphere inside. Baldocchi et al. (1988) stated that the fans within the chamber modify the profile distribution of gases concentration present in field. Steduto and colleagues (2002), investigating the possible impact of the chamber on the CO_2 fluxes of *Lolium perenne*, demonstrated the effect on gas and temperature distribution, without affecting the CO_2 concentration.

Chambers tend to reduce radiation intercepted by the enclosed plants. The degree of attenuation depends on the "*absorbance, transmittance and reflectance of the walls, the type of material used for the transparent walls and their ageing conditions*" (Weinstock, et al., 1982). On the contrary; the chamber walls increase the solar radiation holding long wave radiation. Different chamber materials attenuates the photosynthetic active radiation (PAR) transmittance of 10%, propafilm (BOPP) and polymethyl methacrylate (PMMA), (Murphy, 2007), 12%, polycarbonate, (Steduto, et al., 2002), 20%, PMMA, (Pickering, et al., 1993). Acrylic plastics (e.g., PMMA) are popular materials for leaf chambers because of their high transmission of PAR, but they may have a great influence on the accuracy and stability of CO_2 measurements because of the water and CO_2 absorption (Hunt, 2003).

The small volume and the transparent enclosure are responsible of the greenhouse effect that causes an increase of temperature inside the chamber.

The estimate of the CO_2 fluxes with the SC approach reports the rate of CO_2 accumulation or release over the sample period. However the measurement method inherently alters the normal ambient conditions, changing the CO_2 concentration. Usually photosynthesis of a limited vegetation extension does not alter the atmosphere composition, because of the huge difference of volume scales. The creation of a restricted atmosphere, limited with boundaries, influences photosynthesis and respiration and alters the concentrations gradients between soil, plant and the overlying air.

Photosynthesis absorbs CO_2, reducing its concentration in the chamber headspace. The further decrease of CO_2 inhibits photosynthesis itself. Therefore, during the day the flux of C toward vegetation is underestimated. Plant respiration is less influenced by CO_2 concentration than photosynthesis. The error on the estimate of night fluxes, concerning only respiration, is therefore negligible. During day time the underestimate of photosynthesis induce/cause to a positive C balance, exhibiting a prevalence of respiration.

For assessing the flux with minimal change, the rate of initial concentration should be used rather than the mean rate of the CO_2 concentration over the chamber closure period (Livingston and Hutchinson, 1995). Kutzbach and colleagues (2007) demonstrated that a quadratic regression equation is more appropriate than a

linear equation modelling the $\Delta CO_2/\Delta t$ of single measurements, "*even if closure time were kept short*". However Murphy (Murphy, 2007) developed a method to discriminate between the appropriateness of the equation used for modelling each measure.

Some publication used SC approach to estimate gross (P_g) and net (P_{net}) canopy photosynthesis too (Bremer and Ham, 2005; Kaye, et al., 2005; Lewis, 2010). Bremer and Ham (2005) studied the strategy for partitioning the CO_2 balance in turfgrasses, using measurements from a small surface chamber modified to manipulate chamber pressure. In fact soil respiration (R_s) is sensitive to changes in chamber pressure, hence the creation of overpressure on the headspace suppresses the soil respiration, permitting to estimate P_{net} and canopy respiration (R_c). For the estimate of the P_g, the method involved consecutive measurements carried out with both a sunlit and a shaded chamber. The shaded measures are obtained with an opaque shell that suppress photosynthesis activity and allow the measure of the only respiration component. Subtracting NEE obtained with the shaded chamber ($R_c + R_s$) to NEE measured with the sunlit chamber ($P_g - R_c - R_s$) permits to calculate P_g.

3.1.6 Aims and scope

This work aims to assess the turfgrass C sequestration in a golf course in the province of Verona, northern Italy, through the estimation of C fluxes during one year. The effect of different maintenance intensities on the playing areas of a hole was investigated by a SC approach. An IRGA connected to a transparent chamber was used to estimate the annual net CO_2 exchange between the ecosystem and the atmosphere.

The goals of this study are to 1) to calculate the NEE of the turfgrass and its seasonal variation over one year; 2) to study the relationships between CO_2 fluxes and environmental variables such as air temperature and PAR; 3) to compare the NEE of areas characterized by different degrees of maintenance.

3.2 Materials and methods

3.2.1 Study site

The study was conducted at the Verona Golf Club, located in Sommacampagna (45°24'N, 10°51'E), in the province of Verona (Italy). The golf course includes 18 holes and, together with facilities, occupies 54 hectares (Figure 3.9). The course is situated in a hilly area (140 meters of altitude) and is influenced by the climate of the Lake Garda, which is 15 kilometres away (Table 3.5).

The first 9 holes were established in 1963 and the course was expanded over the following decade. The turfgrass of the experimental field of our case study (18th hole) was established approximately 40 years ago.

Right from the start, the course was conducted with an agronomic and eco-compatible system of management with a low impact on the environment. All the playing surfaces were daily irrigated from March to November, with the exception of days with precipitation higher than 10 mm. The only area that was not irrigated was rough.

Figure 3.9: Site location (image elaborated from Google Earth)

Table 3.5: Monthly means of minimum and maximum temperature, precipitation and relative humidity, related to the last 30 years collected at the meteorological station of Verona-Villafranca (Ilmeteo.it, 2014).

Month	T. min. (°C)	T. max. (°C)	Precip. (mm)	U.rel. (%)
January	-2	5	55	85
February	0	9	48	78
March	3	13	58	73
April	7	17	69	75
May	11	22	85	73
June	15	26	86	73
July	18	29	62	73
August	17	28	88	74
September	14	24	63	76
October	8	18	82	81
November	3	11	75	84
December	-1	6	51	84

Figure 3.10: The 18th hole of the Verona Golf Club (image elaborated from Google Earth).

3.2.2 Experiment description, instrumentation and flux calculations

The study was conducted from August 2012 to September 2013 at the Verona Golf Club. The area used for the surveys was the 18th hole of the golf course, extended for 18,270 m^2, positioned close to the club house and the technical recovery (Figure 3.10).

The course of the hole includes different playing areas, characterized by different species composition (Figure 3.11) and management intensity (Table 3.6). In descending order from the high maintained playing areas, we define green, tees, collar, fairway, semi-rough, mount and rough.

The degree of management of each playing area was determined collecting data regarding cultural operations during the surveys period. Those data, summarized in Table 3.6, were provided by the superintendent of the Verona Golf Club, who monitored all the cultural operations effectuated in the different playing areas during the period of the present study.

Mowing is the most important and frequent maintenance activity for turfgrass. The surfaces with higher playing pressure are mowed more frequently than the others. The green of the case-study hole, for example, was mowed almost every day during growing season, while fairways was mowed three times per week (Table 3.6). Mowing activities was interrupted only from December to February. Applications of selected fertilizers for turf were constantly provided from March until September on green (total of 56 kg, 1058 kg/ha), tees (total of 20 kg, 385 kg/ha) and fairway (total of 474 kg, 710 kg/ha). Pesticides applications were focused on the most

sensitive areas (green and tees). Fairways turfgrass was verticutted at the end of March, while green received the same treatment three times between April and May. Topdressing operations followed verticutting. Fairway was over-seeding with cool-season grasses at the end of August.

Table 3.6: Cultural operations count from September 2012 to August 2013.

Playing area	Mowing	Fertilization	Agrochemical application	Verticutting	Top-dressing	Over-seeding
GREEN	225	13	12	3	9	-
TEE	127	6	9	-	-	-
COLLAR	113	-	-	-	-	-
FAIRWAY	106	6	-	1	2	1
SEMIROUGH	91	-	-	-	-	-
MOUNT	74	-	-	-	-	-
ROUGH	42	-	-	-	-	-
Total	**778**	**25**	**21**	**4**	**11**	**1**

Within the course, 20 NEE measurement points were marked with iron plates (Figure 3.12) positioned in different playing areas. These areas have been then grouped in 3 categories according to their degree of maintenance: high intensity (HI), including tees, green and collar, medium intensity (MI), including fairway and semi-rough, and low intensity (LI), including rough and mount. HI, MI and LI had respectively 4, 12 and 4 measurement points (Table 3.7). The number of measuring points repeated for playing areas was proportional to the width of their area within the hole.

Table 3.7: Subdivision of playing areas in 3 maintenance categories: high intensity (HI), medium intensity (MI) and low intensity (LI). Measuring points grouped per area and surfaces.

Management category	Playing area	Measurem. points	Surface (ha)	Percentage of 18th hole
HIGH INTENSITY	Green	1	0.0541	3.0%
	Tee	2	0.0512	2.8%
	Collar	1	0.0120	0.7%
MEDIUM INTENSITY	Fairway	9	0.6675	36.5%
	Semirough	3	0.2800	15.3%
LOW INTENSITY	Mount	1	0.0120	0.7%
	Rough	3	0.7500	41.1%
Total measuring points		**20**	**1.8268**	**100%**

A meteorological station (Figure 3.13) was positioned within the golf course (Lat. 45°23'37.87"; Lon. 10°49'19.53''; Alt. 131 meters). The station is equipped with the following sensors: air temperature (-30 ÷ +70 °C), relative humidity (0 ÷ 100% U.R.), precipitation (res. 0.2 mm), wind velocity (0 ÷ 50 m/s), solar radiation (0 ÷ 1500 W/m^2) and evaporation (0 ÷ 100 mm).

Whole-canopy gas exchange measurements were performed with a portable IRGA (EGM-4, PP Systems, UK) equipped with a canopy chamber (CPY-2, PP Systems, UK) (Figure 3.13) characterized by the following measures:

-14.5 cm height x 14.6 cm diameter;

- 167 cm^2 of exposed area;

- 2425 cm^3 of volume.

The chamber, designed for closed system measurement of canopy CO_2 fluxes, is transparent and fitted with an air mixing fan and sensors for measurement of PAR and air temperature. A tapered stainless steel edge (10 mm) is attached along the base of the chamber to push into the soil and improve the seal between the chamber and the soil surface (Figure 3.13).

For our study we considered the CO_2 flux values collected at the end of the each measurement that lasted between 32 and 40 seconds.

The surveys were carried out every 2 weeks, for a total of 23 measurement days over one year. Each survey consisted in 5 measures taken at 5.30, 9.30, 14.00, 18.00 and 22.00, for each of the 20 measurement points.

Every single measurement lasted 40 seconds in the first 6 surveys, then 32 seconds for the rest of the study. In order to account for the effect of the chamber on the headspace parameters, temperatures inside the chamber were recorded at the beginning and at the end of each measurement. The variation of pressure within the chamber was not considered influential for the CO_2 flux measurement.

The chamber was positioned every time in the same positions, following a random order between one point and the other. The instrument (IRGA and chamber), has an inbuilt data logger, and provided the value of CO_2 flux expressed as $\mu molm^{-2}\ s^{-1}$ CO_2. The basic formula adopted for computing CO_2 flux (J_c) is:

$$J_c = \rho_m \frac{V}{A} \frac{\Delta CO_2}{\Delta t} \qquad \textbf{(eq. 3.6)}$$

where J_c is the CO_2 flux($\mu mol\ m^{-2}\ s^{-1}$), ρ_m is the molar density of air (μmol air m^{-3} air) calculated from the ideal gas law, V is the chamber volume (m^3 air), A is the chamber area (m^2) and $\Delta CO_2/\Delta t$ is the rate changes over time of CO_2 concentrations within the chamber ($\mu mol\ CO_2\ \mu mol^{-1}$air s^{-1}) (Murphy, 2007).

Together with the CO_2 concentration (ppm), and flux, the air temperature (°C) and the PAR (µmol m^{-2} s^{-1}) inside the chamber were recorded by chamber-integrated sensors.

Daily CO_2 flux values for each measurement point were obtained from the integration of the 5 measurements with the trapezoidal rule, by approximating the region under the graph of the function fitting the data as trapezoid and calculating its area. The 5 measurement times were chosen in order to be representative of the different light and temperature conditions during a day. The estimated daily NEE was then considered representative of the period between one measurement day and the following (approximately 15 days, except in the beginning of October and at the end of December). The same approach was then used to calculate the annual NEE.

Statistical Analysis

Normality of the NEE annual data was tested with the Shapiro-Wilk test. One-way analysis of the variance (ANOVA) with Statgraphics Centurion XV (StatPoint Inc., USA) has been carried to test the effect of the management intensity on the seasonal and annual cumulated values of NEE. A multiple comparison procedure was then used to analyse statistically significant differences among means using the Tukey's honest significance test ($p<0.05$).Homogeneity of variance was checked using Levene's test before analysis. The significance of the relationships between ecosystem respiration and temperature during the night and between photosynthesis and PAR during the day, have been tested by linear regression analysis with Table Curve 2D, v4.07 (Systat Software Inc., USA).

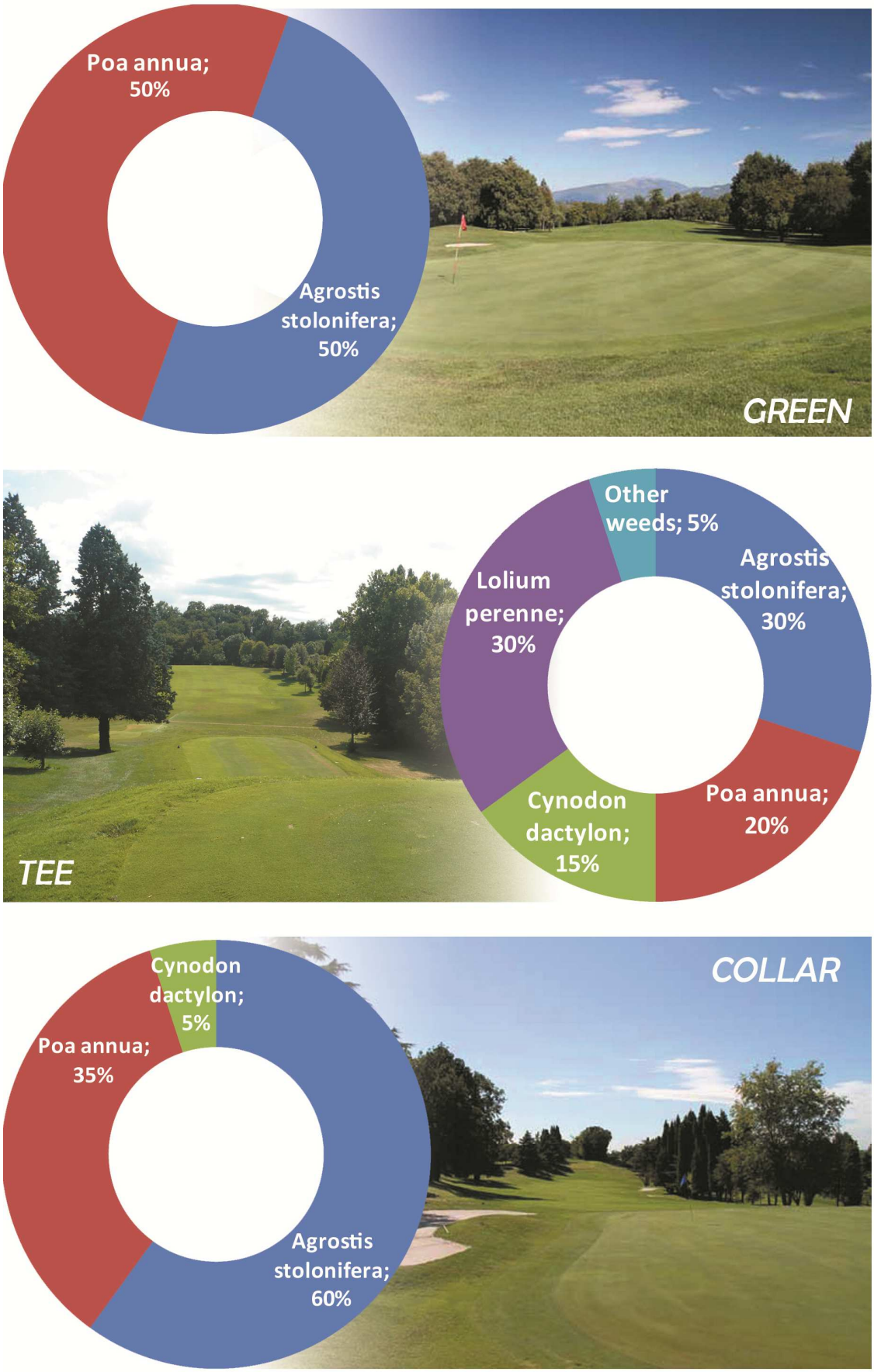


Figure 3.11a: Species composition of each playing areas: green, tees and collar.

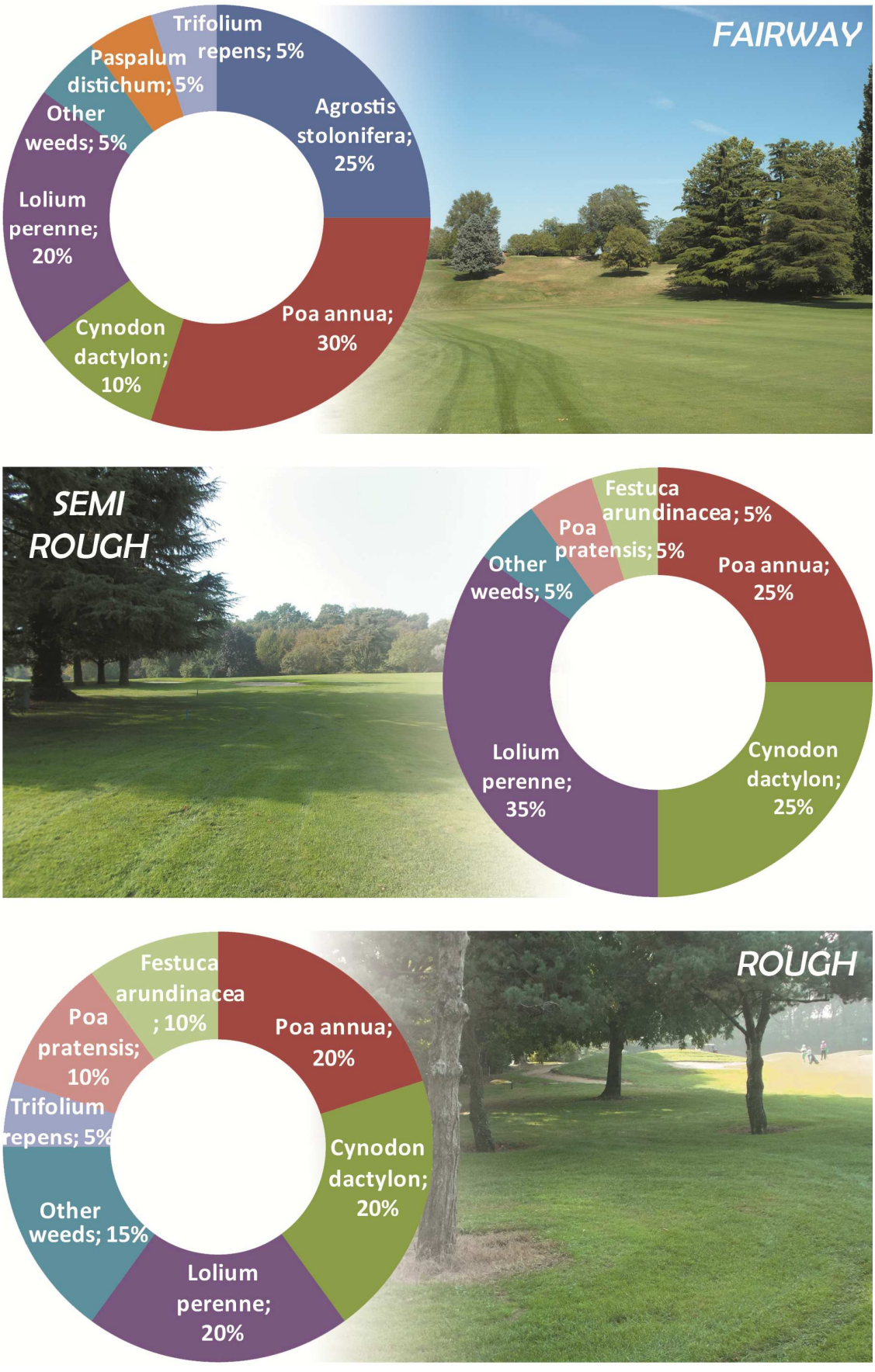


Figure 3.11b: Species composition of each playing areas: fairway, semi-rough and rough.

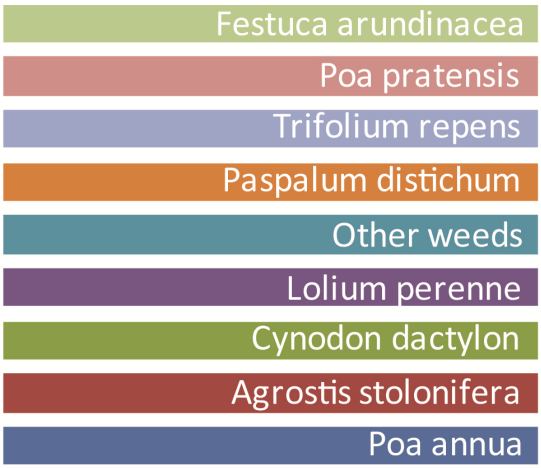


Figure 3.11c: Legend.

Figure 3.12: Map of the 18th hole: playing areas and location of 20 iron plates, identified with GPS system. High intensity management category (red) includes tees (2), green (1) and collar (1). Medium intensity management category includes fairway (8) and semi rough (3). Low intensity management category includes rough (3) and mount (1). In brackets number of measurement points.

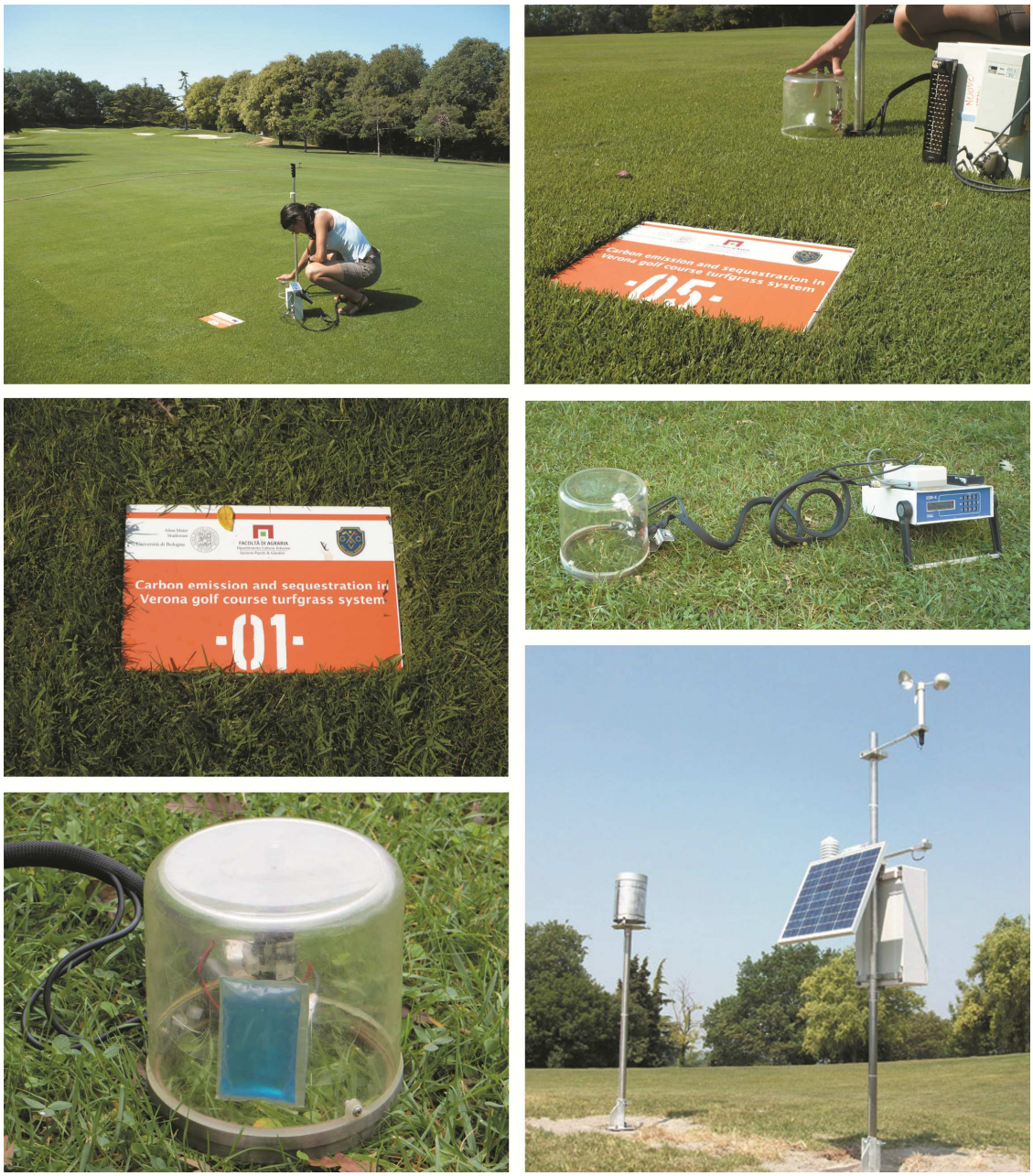


Figure 3.13: Above: measurement during field survey; centrally: numbered iron plate marking NEE measurement point and portable IRGA (EGM-4, PP Systems, UK); below: canopy chamber CPY-2 (PP Systems, UK) and meteorological station.

3.3 Results

3.3.1 Meteorological and phenological seasonality in the studied turfgrass

Figure 3.14 reports mean monthly values of air temperature and cumulated precipitation during the studied period. Air temperatures ranged from -4.4°C (December 2012) to 37.2°C (July 2013). The cumulated precipitation was 2356 mm and resulted abundant during autumn and springtime, exceeding 100 mm per month in September, October and November 2012 and March, April and May 2013, with an extraordinary peak on May 2013 (243 mm)(Table 3.5) and eventually reached his minimum values during summer 2013, with June, July and August having 50, 12 and 40 mm of rain, respectively (Figure 3.14). July and August 2013 fall in the aridity period, according to Walter postulate (Walter, 1960).

Optimal temperatures for cool season-grasses photosynthesis range between 15 and 24°C (Croce, et al., 2006), permitting the turf to thrive during spring and autumn at our latitudes, and to remain green during the summer and winter except when temperatures fall below 0 °C. Considering the trend of meteorological data during this study, turfgrass growing season in Golf Club Verona lasted from 15 August to 15 November 2012 (autumn) and from to 15 March to 15 July 2013 (spring). Turf remained green during the month from 15 July to 15 August (summer), tolerating drought thanks to the irrigation, and went to dormancy from 15 November to 15 March (winter).

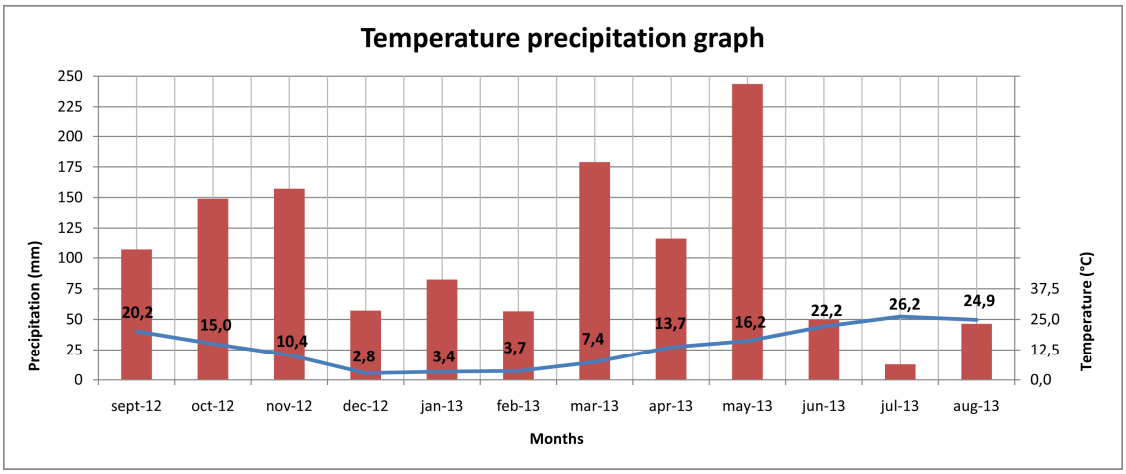


Figure 3.14: Monthly mean air temperature and cumulative rainfall during the experimental period. Red bars indicate monthly precipitations, blue line indicate monthly temperature variations.

3.3.2 Net ecosystem exchange measurement

Figure 3.15 shows the daily average NEE trends and PAR values for each maintenance category, as measured in the 5 sampling times in 4 representative days of the different seasons, while Table 3.8 shows the seasonal variability on single daytime and nigh time measurements.

In springtime and autumn diurnal values of NEE were similar both in the maxima and in the minima, the latter representing the highest diurnal uptake rates observed in the whole period (-2.46 and -2.13 µmol CO_2 m^{-2} s^{-1} respectively). Nigh time respiration was however higher in springtime and summer than in autumn (4.78 compared to 2.75 µmol CO_2 m^{-2} s^{-1}). The maximum and minimum diurnal values during summer were closer to zero (-1.49 and 1.97 µmol CO_2 m^{-2} s^{-1} respectively) compared to spring time and autumn. During the winter, the diurnal values of NEE were always negative while the nocturnal respiration obviously showed the lowest values (Table 3.8).

During the day the relationship between CO_2 flux (µmol m^{-2} s^{-1}) and PAR (µmol m^{-2} s^{-1}) was well described by linear functions. Figure 3.16 shows the relationship between NEE and PAR in 4 days representative of different season. The C uptake of the system (negative NEE) increases in absolute value with increasing PAR, though the slope of the regression lines varies in the different season. The 27th of July, notwithstanding the high PAR values, NEE was often positive, indicating that C losses prevailed over uptakes.

Net ecosystem exchange sampled with PAR equal to zero (nocturnal and early morning measurements) represents the ecosystem respiration ($R_{eco}=R_a+R_h$) whose relation with air temperature inside the chamber is well described by an exponential function. Figure 3.17 reports the relationship between air temperature inside the chamber and CO_2 flux all over the year for each management category. The mean temperature sensitivity of respiration (Q10) however, increased with increasing intensity management with values comprised between 3.37 for LI and 4.46 for HI, the latter significantly higher than the former ($p<0.05$) (Figure 3.18).

Figure 3.19 shows the annual trend of NEE for each category of maintenance intensity. Net ecosystem exchange ranged between -9.41 and +70.53 mg C m^{-2} d^{-1}. In the period between 16th November 2012 and 27th March 2013 the mean daily NEE was -2.05 mg C m^{-2} d^{-1}, with values ranging from -9.41 to 7.06 mg C m^{-2} d^{-1} for all the categories. In April 2013 values of NEE quickly increased, reaching a peak of CO_2 emission comprised between 30 and 50 mg C m^{-2} d^{-1}. In May NEE was again close to zero then gradually increased during the summer peaking in July at values comprised between 40 and 70 mg C m^{-2} d^{-1}. The sequestrated C during the daytime resulted slightly greater than the emitted C during the night in late autumn and winter, whereas from spring to early autumn, the C via autotrophic and

heterotrophic respiration exceeded that sequestered by photosynthesis, hence NEE trend during this period assumed positive values.

All the maintenance categories areas showed similar NEE trends (Figure 3.19), however the mean daily CO_2 fluxes were different according to the intensity of maintenance and the season (Table 3.9). Mean daily values of NEE in HI were lower than those in MI and LI in spring, summer and autumn, whereas in winter the lower mean daily value was in MI and, together with the HI value of the same season, was the only seasonal negative rate. The higher CO_2 fluxes values were reached in summer for all categories (Table 3.9). The results of the ANOVA show how the cumulative NEE is affected differently by management intensity along the year (Table 3.10), with MI and HI having significant lower NEE values than LI ($p<0.05$) in autumn and winter, while in summer HI was significantly lower than MI but not than LI. No significant differences between management intensities were found in springtime

This different seasonality of the NEE resulted as well in its annual integration (Figure 3.20). The annual NEE showed a decrease (therefore an increase in the annual C budget) with the increase of the intensity of management, with HI significant lower ($p<0.05$) than LI while MI was not significantly different from both LI and HI (Figure 3.20).

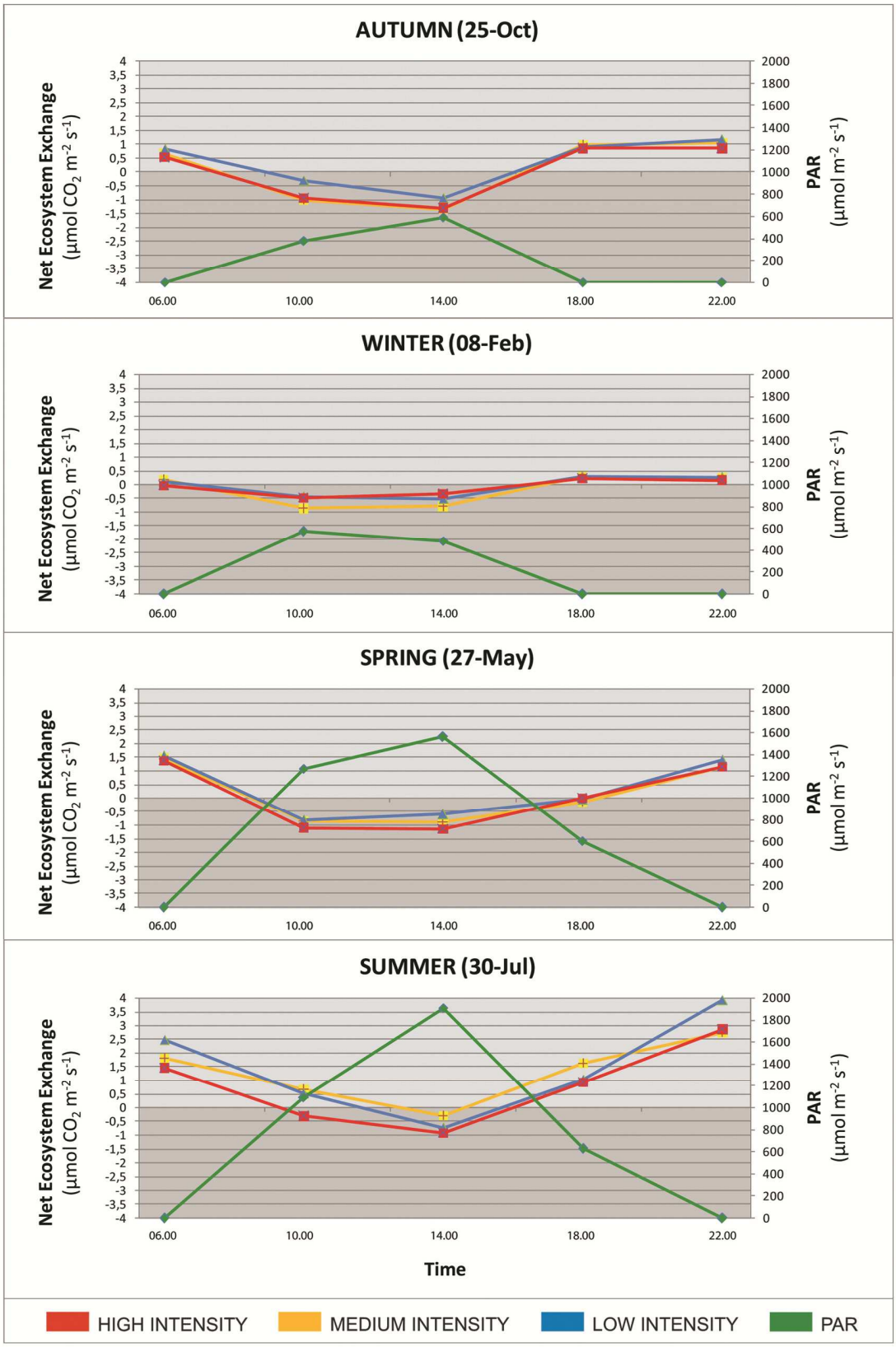


Figure 3.15: Daily trend of photosynthetically active radiation (PAR) and average net ecosystem exchange (NEE) for each maintenance category on four representative days in autumn (25th October), winter (8th February), spring (27th May) and summer (30th July). Maintenance categories are HI (high intensity), MI (medium intensity) and LI (low intensity), n= 4 for LI and HI, 12 for MI. PAR values are measured from the meteorological station.

Table 3.8: Diurnal and nocturnal extremes values of net ecosystem exchange (NEE) divided for turf phenological season. NEE values are expressed in µmol CO_2 m^{-2} s^{-1}. Maintenance categories are HI (high intensity), MI (medium intensity) and LI (low intensity).

turf season	date	NEE (µmol CO_2 m^{-2} s^{-1})		category
winter	16/11/12 (survey: 10.00)	-2.05	daily min	LI (mount)
15-Nov ÷ 15-Mar	16/11/12 (survey: 10.00)	-0.01	daily max	LI (rough)
(4 months)	1/3/12 (survey: 22.00)	0.72	nocturnal max	MI (semi-rough)
spring	11/4/12 (survey: 14.00)	-2.46	daily min	MI (fairway)
15-Mar ÷ 15-Jul	5/7/12 (survey: 09.30)	3.72	daily max	HI (tee)
(4 months)	4/7/12 (survey: 22.00)	4.78	nocturnal max	MI (fairway)
summer	13/8/12 (survey: 14.00)	-1.49	daily min	LI (mount)
15-Jul ÷ 15-Aug	30/7/12 (survey: 18.00)	1.97	daily max	MI (semi-rough)
(1 months)	30/7/12 (survey: 22.00)	4.36	nocturnal max	LI (mount)
autumn	20/9/12 (survey: 14.00)	-2.13	daily min	LI (mount)
15-Aug ÷ 15-Nov	4/10/12 (survey: 18.00)	2.85	daily max	LI (mount)
(3 months)	4/10/12 (survey: 22.00)	2.75	nocturnal max	MI (semi-rough)

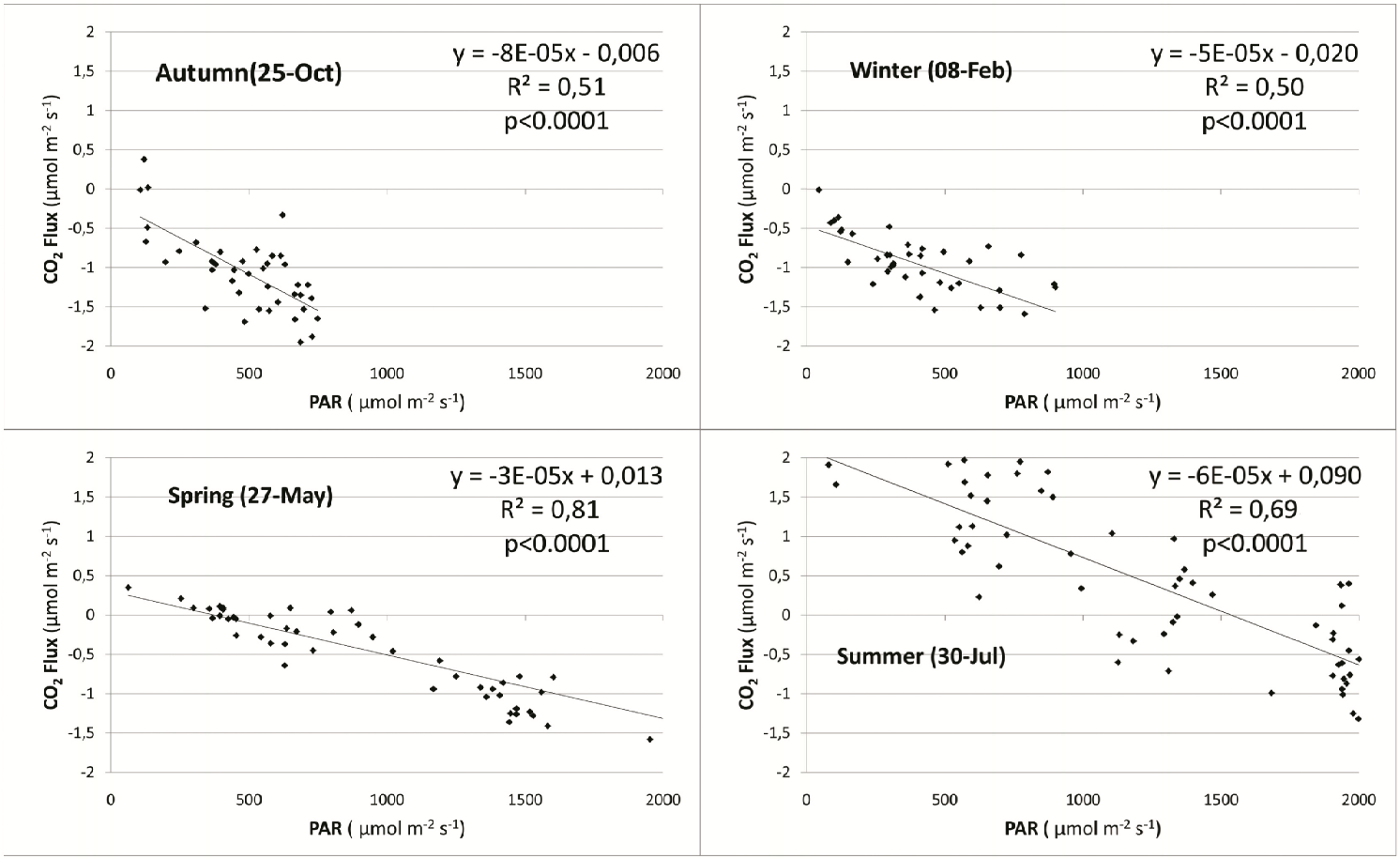


Figure 3.16: Relation between PAR (µmol m^{-2} s^{-1}) and CO_2 flux (µmol m^{-2} s^{-1}) on four representative days in autumn (25th October), winter (8th February), spring (27th May) and summer (30th July).

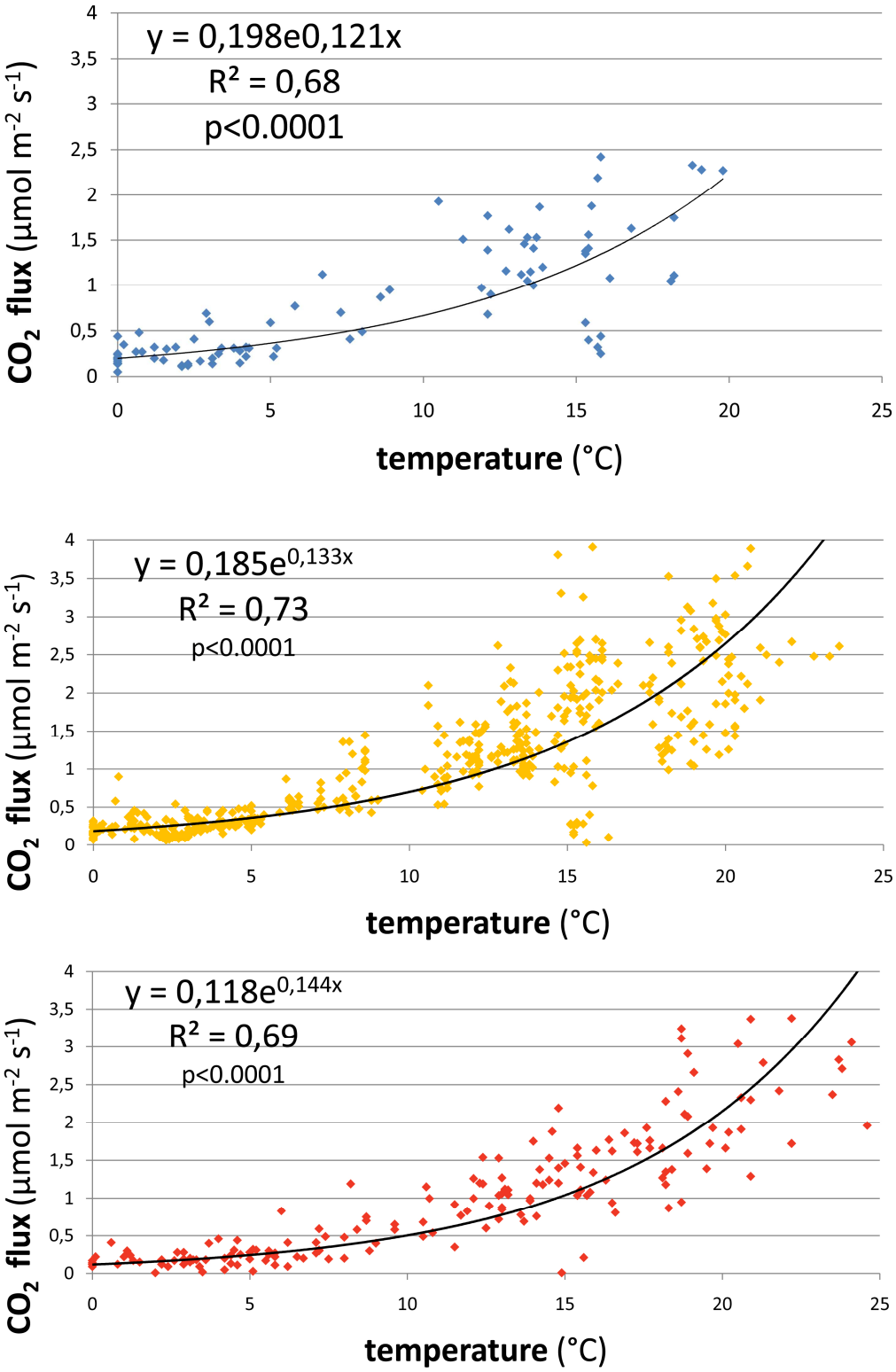


Figure 3.17: Relationship between ecosystem C flux and air temperature inside the chamber with PAR=0 (nocturnal measurements). All the measurement values are divided for maintenance categories: low intensity (LI, blue), medium intensity (MI, yellow) and high intensity (HI, red).

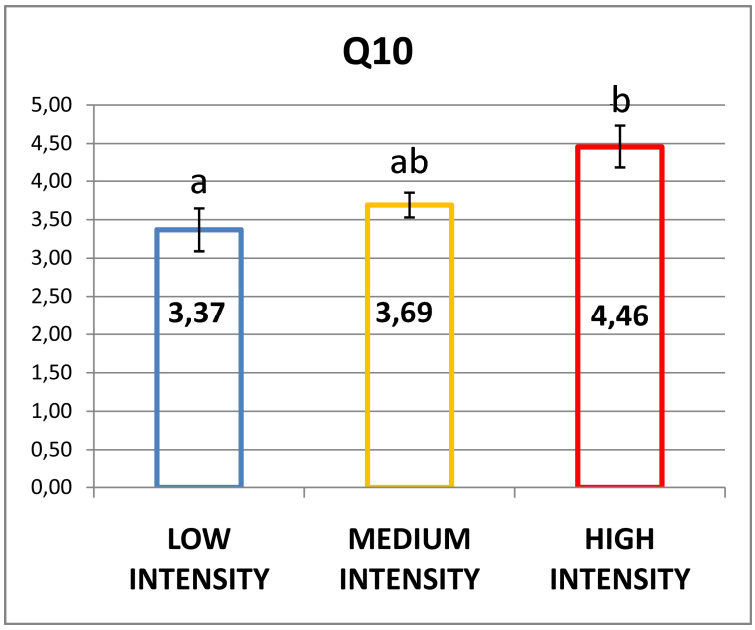


Figure 3.18: Average Q10 for each maintenance category. Bars are SE of the mean, n=4 for LI; 12 for MI; 4 for HI. Different letters indicate statistically significant differences, $p<0.05$.

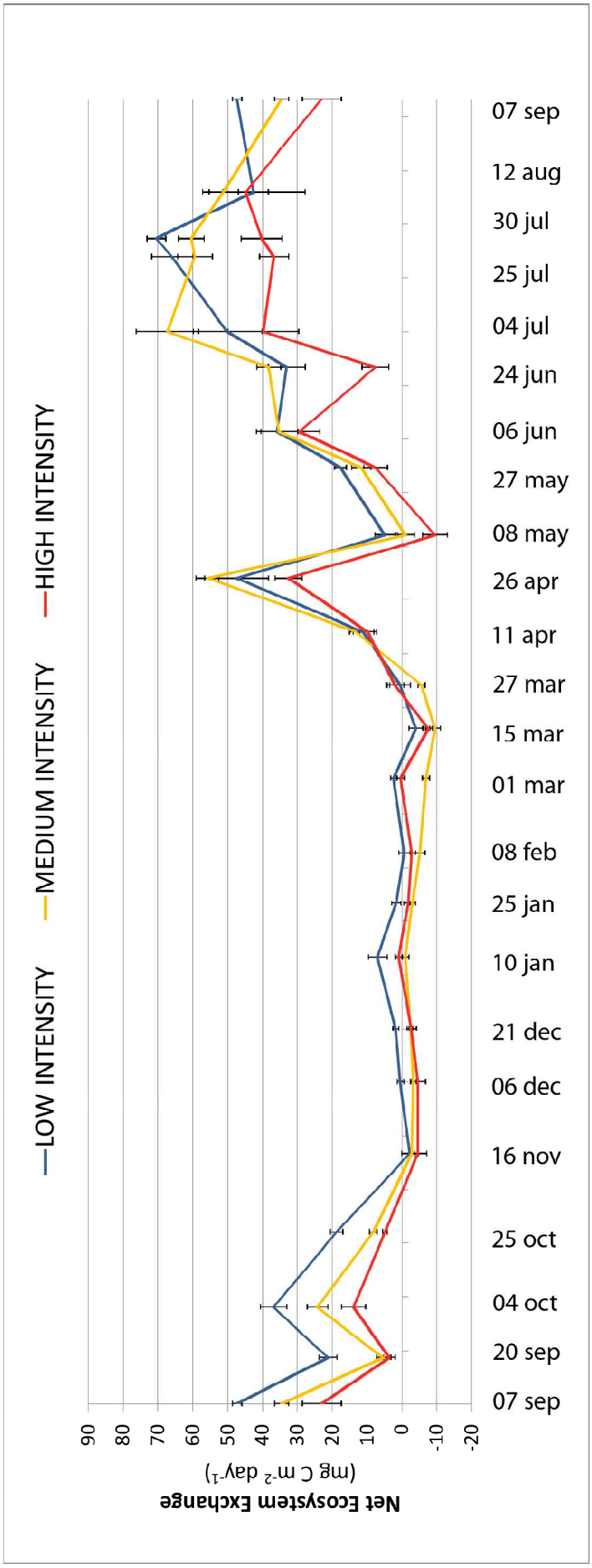


Figure 3.19: Annual trend of daily net ecosystem exchange (NEE) of turfgrass under different management intensities. Bars are SE of the mean, n= 4 for LI and HI, 12 for MI.

Table 3.9: Average daily net ecosystem exchange (NEE) for turfgrass managed at different maintenance degree, in the four turf phenological seasons. Maintenance categories are HI (high intensity), MI (medium intensity) and LI (low intensity). Values are mean ± SE of the mean, n= 8 for winter and spring, 3 for summer and 4 for autumn.

Average daily NEE (mg C m^{-2})			
turf season	***LI***	***MI***	***HI***
winter (8 surveys)	0.86±1.60	-4.21±0.97	-2.78±1.57
spring (8 surveys)	25.15±5.11	26.91±3.65	14.99±4.43
summer (3 surveys)	59.73±7.77	57.10±4.30	40.68±5.51
autumn (4 surveys)	31.10±2.35	18.15±2.05	11.44±2.84

Table 3.10: Average cumulative net ecosystem exchange (NEE) for each turf phenological season, at different maintenance degree, obtained from the integration of daily NEE by the trapezoidal rule. Maintenance categories are HI (high intensity), MI (medium intensity) and LI (low intensity). Values are mean ± SE of the mean, n= 8 for winter and spring, 3 for summer and 4 for autumn. Different letters indicate statistically significant differences, $p<0.05$. All statistical analysis was carried out using ANOVA.

Average cumulative seasonal NEE (mg C m^{-2})				
	Autumn	Winter	Spring	Summer
LI	2.24±0.21 **b**	0.36±0.11 **b**	2.35±0.33 **a**	2.50±0.15 **ab**
MI	1.43±0.11 **a**	-0.40±0.08 **a**	2.35±0.26 **a**	2.40±0.18 **b**
HI	1.04±0.16 **a**	-0.27±0.11 **a**	1.26±0.20 **a**	1.56±0.17 **a**

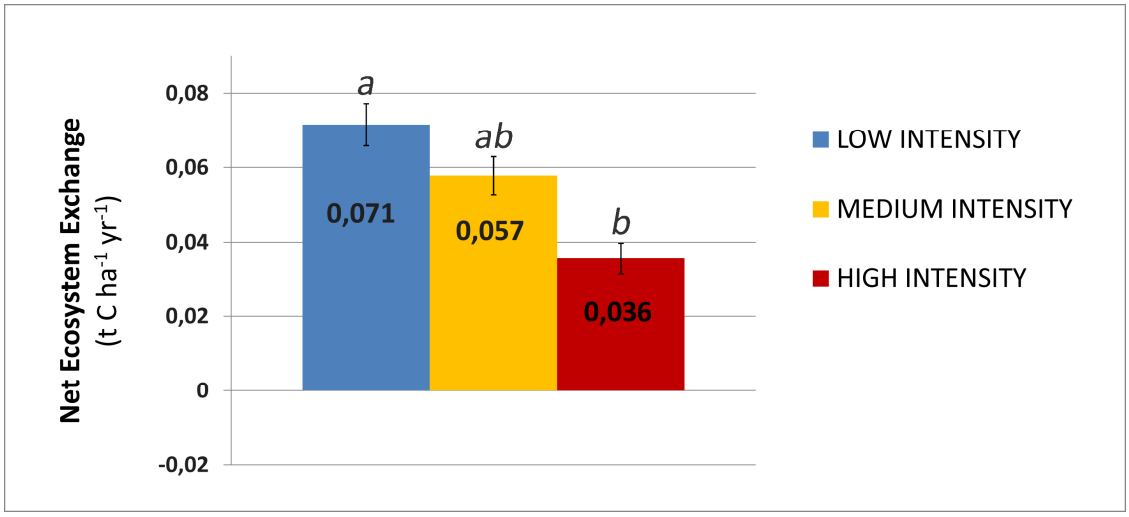


Figure 3.20: Annual cumulative net ecosystem exchange (NEE) of turfgrass under different management intensities. Values are means, n=4 for LI; 12 for MI; 4 for HI. Bars are SE of the mean. Different letters indicate statistically significant differences, p<0.05.

3.4 Discussion

3.4.1 Criticism of the measuring system

The major criticisms related to the utilization of the small enclosures approach to measure NEE have been analysed in the introduction. Annual estimates of NEE can be influenced mainly from two types of errors: 1) connected to the measuring system and 2) related to the measurement frequency, timing and duration overtime.

As discuss by several authors (Steduto, et al., 2002; Bremer and Ham, 2005; Kutzbach, et al., 2007) to enclose the vegetation in a chamber may cause a perturbation of micro environmental variables such as pressure, incoming radiation, temperature and CO_2 concentration in the chamber headspace. This together with possible leaks in the closed system may alter the natural CO_2 flux of the system.

Despite some studies indicate pressure variation as an important factor affecting soil respiration (Fang and Moncrieff, 1998) and NEE estimates with transparent chambers in turfgrass (Bremer and Ham, 2005), other studies proved that short closure times for transparent chambers caused only negligible error in CO_2 flux calculation (Steduto, et al. 2002). In the present study this variable was not measured or controlled and the disturbance caused by possible changes in pressure were considered negligible.

A reduction of 10-20% of the incoming radiation at canopy level, caused by the chamber walls has been reported in previous studies (Pickering, et al., 1993; Steduto et al. 2002; Hunt, 2003; Murphy, 2007) together with an increase in the diffused component of the solar radiation, which can cause more effective light

penetration inside the enclosed canopy. The influence of such disturbances on plant photosynthetic activity during the measurements could have had different sign and was considered negligible in our system.

Concerning greenhouse effect, literature reported chamber temperature increases between the start and the end of the measurement of 3°C (Wagner and Reicosky, 1992; Murphy, 2007; Müller, et al., 2009), 1-2°C (Steduto, et al., 2002), 0.7°C (Lewis, 2010). As tested by previous studies, we reduced as much as possible the measurement period in order to minimize the greenhouse effect. In our measurements the difference of temperature between inside and outside the chamber, calculated on the period with higher average environmental temperatures was in average +0.47±0.76°C (SE, n=700), which is comparable with what reported by previous studies.

Warmer conditions within a chamber system can result in an underestimation of the CO_2 uptake rate compared to micrometeorological methods (Dugas, et al., 1999). A temperature increase could affect NEE both at photosynthesis level and respiration level. In a similar system, the build-up of leaf or/and air temperature has been shown to be partially responsible of the underestimate of NEP, up to 10% (Wagner and Reicosky, 1992). An inhibition on photosynthesis likely occurs when the CO_2 concentration inside the chamber quickly decrease as a consequence of the fixation by the plants. The change of CO_2 concentration from standard conditions altered the concentration gradients beneath the chamber, making CO_2 concentration a limiting factor that altered photosynthesis and, therefore, C uptake. In other words, the measurement method itself alters the measures (Kutzbach, et al., 2007). This combined effect of temperature and CO_2 concentration likely increased during the warm season, enhancing the underestimation when the photosynthetic activity is higher.

The change of CO_2 concentration and temperature within the chamber headspace can affect both autotrophic and heterotrophic respiration, however soil respiration have a slower kinetic than photosynthesis and, during such short measurement time the change of conditions do not influence microorganism metabolism (Curiel Yuste, et. al, 2007). Plant autotrophic respiration in the other hand, likely increased during measurements, resulting in an increase of respired CO_2 which lead to an overestimation of R_{eco}. The methodological errors related to the underestimation of photosynthesis and to the overestimation of respiration had the same sign in terms of NEE and were added each other.

3.4.2 Net ecosystem exchange of the turfgrass

The high precipitations values recorded in the studied period, together with the irrigation provided in all the hole surfaces, excepted for rough, likely created optimal soil moisture content for turfgrass during the whole year. As a consequence, the turf has followed the typical growing pattern for cool-season grasses, which consist in two distinct growing periods one during the autumn (15 of August to 15 of November) and one during the springtime (15 of March to 15 July) when daily optimal temperatures were reached (Figure 3.21).

Figures 3.15 and 3.19 highlight the seasonality of the NEE in the turfgrass. The trends seem to follow more the temperature trend (Figure 3.21) than the physiological activity of the turf; however interaction between the two likely occurs. During the winter the NEE values close to zero (Table 3.9), but always negative, indicate a prevalence of the photosynthesis over the respiration. So far, only Zhou et al. (2012) studied the role of turfgrass in C balance during dormancy season, measuring CO_2 fluxes during the transition from winter to spring in a temperate urban lawn characterized by a warm season grass (*Zoysia japonica*) in Beijing, China. In their study the NEE had a wider range (from -2.39 to +2.87 µmol CO_2 m^{-2} s^{-1}) compared to cool-season grass of our study (Table 3.8), with an average CO_2 rate of +0.13 µmol m^{-2} s^{-1} between mid-January and the end of March. These values showed that, despite the different metabolism, the behaviour of warm- and cool-season grasses in winter was similar. The acclimation strategies adopted by the C4 plants, consisting in maintaining photosynthesis active also in the winter months, allowed some periods of CO_2 uptake (Zhou, et al., 2012).

Pataki (2006) in a single winter daily survey, found opposite results in plots of *Festuca arundinacea* on Southern California (United States), measuring a NEE value of -6.38±1.41 µmol CO_2 m^{-2} s^{-1}, significantly lower than our results.

Daily values of NEE measured in spring and autumn are similar (Table 3.8). In both cases maximum C accumulation was sampled at 14.00 in correspondence to maximum PAR conditions. Net uptake of CO_2 during springtime has been previously reported by Lewis (2010) for *Festuca arundinacea* and *Lolium perenne* turfgrasses in Kansas (USA) at the same moment of the day in June, although 4 times higher than those measured in the present study (Table 3.11). Despite NEE is usually negative on the surveys carried out in the morning and early afternoon, (e.g. Figure 3.15), average daily NEE in spring and autumn resulted positive (Table 3.9).

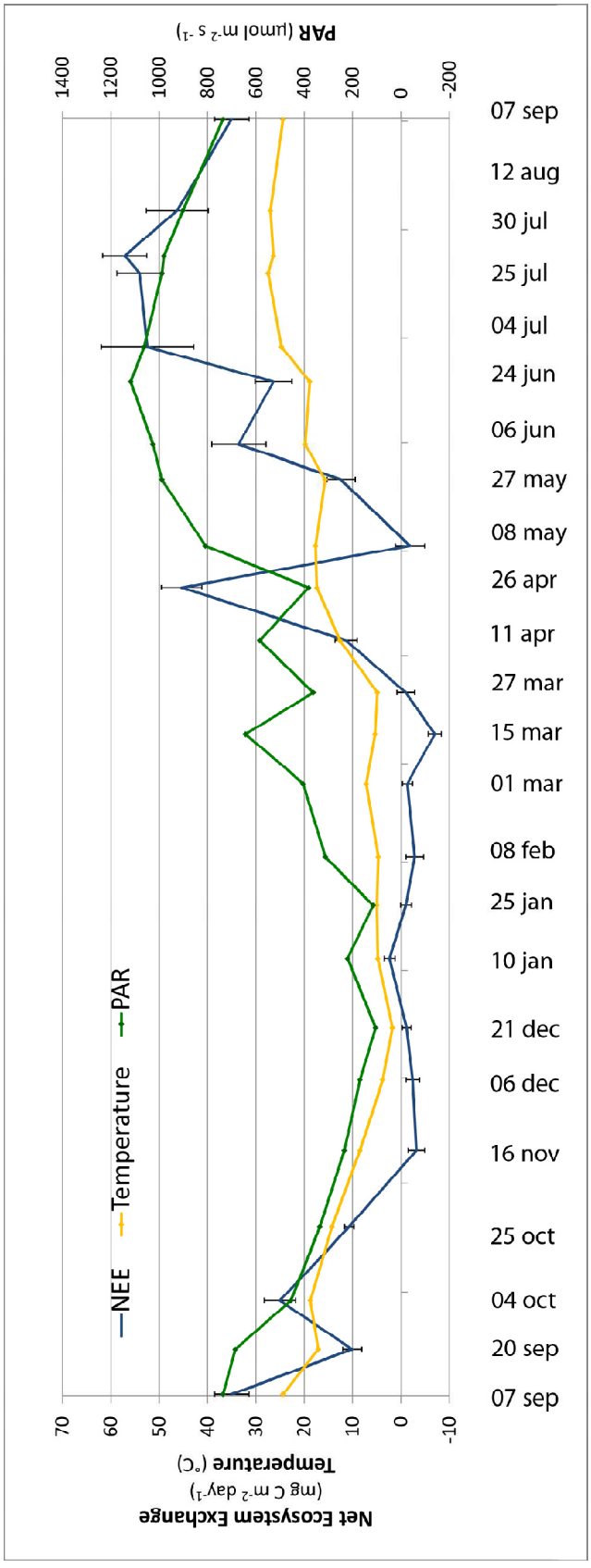


Figure 3.21: Annual trend of daily net ecosystem exchange (NEE) of turfgrass in relation to temperature and PAR. Bars on NEE trend are SE of the mean, n= 20.

Table 3.11: Turfgrass gas exchange measurements as reported in the literature. All the values are expressed in µmol CO_2 m^{-2} s^{-1}. P_{net} and P_g are related to fluxes toward the atmosphere while net ecosystem exchange (NEE) concerns fluxes towards soil. Data with * represent the maximum flux value of the season. (*) and (**) are collected respectively on DOY 217 and 262.

SMALL CHAMBER GAS EXCHANGE							
site	tipology	instrumentation	NEE	Pnet	Pg	season	pubblication
			µmol CO , m^{-2} s^{-1}				
Irvine (California), United States	*Festuca arundinacea*	LI-COR + chamber	-6.38±1.41 -7.18±0.47			sum. win.	Pataki, 2006
Quebec City Canada	different urban lawns	gas chromatogr. + chamber	-1.29 ÷ +14.32			spr. sum. aut.	Allaire, et al., 2008
Beijing, China	urban lawn (*Zoysia japonica*)	IRGA + automatic chambers	+0.13			win.	Zhou, et al., 2012
Victoria, Australia	different urban lawns (*Microlaena spp., Festuca spp., Lolium spp.*)	IRGA + automatic chambers	+4.60 ÷ +18.70			sum. aut.	Livesley, et al.,2010
Manhattan (kansas), United States	*Festuca arundinacea, Lolium perenne*	IRGA + chamber	-12.3			spr.	Lewis, 2010 (ch.3)
Wooster (Ohio), United States	*Festuca arundinacea* and *Lolium perenne* , colonized by fungal endophyte	LI-COR + chamber	+3.15 ÷ +15.14			sum.	Singh, 2007
Manhattan (kansas), United States	*Festuca arundinacea* *Lolium perenne* *Poa pratensis*	LI-COR + chamber, Bremer and Ham method	+9.62 +10.16 +9.41	14.33 14.05 11.54	17.37 17.33 14.07	spr. sum. aut.	Bremer and Ham, 2005
Manhattan (kansas), United States	*Cynodon dactylon* *Zoysia japonica* *Festuca arundinacea* *Poa pratensis*	LI-COR + chamber, Bremer and Ham method		25.2 (*) 18.4 (*) 20.7 (**) 6.4 (**)	43 (*) 33 (*) 27 (**) 20 (**)	sum. aut.	Lewis, 2010 (ch.5)
Chile	*Festuca arundinacea* (cv. Cochise and Bingo) *Cynodon dactylon*	IRGA + chamber	-15 -75			spr. sum. aut.	Acuna, et al., 2012

Water availability may have affected the magnitude of the soil respiration, as suggested by some studies (Fortin, et al., 1996; Wagai, et al., 1998) influencing the net CO_2 flux of the turf. The influence of moisture content on soil CO_2 is complex because it affects several components of the plant-soil system (roots, microbes, gas transport through the soil, etc.) (Fang and Moncrieff, 2001).

In our turfgrass the availability of subtle-broken OM (turfgrass clippings) and water (precipitation and irrigation), combined to the warm temperatures, could have influenced the CO_2 fluxes, enhancing diurnal and nocturnal soil respiration during the whole growing season. The peak in NEE observed in the two surveys carried out in April (Figure 3.19) is concomitant with the start of the physiological activity of the turf and with a sudden increase of 10 °C in the air temperature after the winter period. Although the diurnal CO_2 uptake was among the higher registered in the period, the respiration during the night was twice as high (Table 3.8) leading thus to a net daily source of C.

Figure 3.21 shows also a low average PAR value registered on the 26thof April, when the NEE peaked, which could have reduced the photosynthetic activity. An American study (Bremer and Ham, 2005) confirmed that, despite the diurnal uptake of CO_2, expressed as P_g and P_{net}, resulted positive during the growing season, NEP were negative due to the high soil respiration (Table 3.11).

Despite the positive daily and seasonal NEE registered during the summer (Figure 3.19 and Table 3.10), the availability of water and the high temperatures maintained active photosynthesis also in that period, with the highest CO_2 uptake values measured around midday (Table 3.8), and similar values are shown at the same time in Figure 3.15.These results suggest that photosynthesis was probably active during all the day, in agreement with Lewis (2010), but soil and canopy respiration exceeded it, making daily average NEE positive. One reason for that could be a decrease in diurnal photosynthesis, which was more inhibited by higher temperatures compared to respiration.

The C source behaviour of the system (Table 3.11) was constant for all the growing season, lasting from the beginning of spring to the end of autumn, with higher peaks in April and July (Figure 3.19). The only data that did not follow this trend were collected on the 15th of March and on the two surveys of May when NEE considerably decreased (Figure 3.19 and 3.21). The extraordinary precipitation occurred in March (180 mm) and May (244 mm) (Figure 3.14) may have conditioned physiological processes. In particular, it is well known that water logging influences the diffusivity of soil pores, reducing oxygen availability for microbial activity (Fang and Montcrieff, 2001). Measurements data suggested that water saturation affected more respiration than photosynthesis, significantly decreasing the high rate of the first one.

The trend of NEE shown in Figure 3.19 were partially confirmed by a Canadian study (Allaire, et al., 2008), that measured CO_2 fluxes through weekly surveys from May to November in two urban lawns subjected to different maintenance practices. Positive CO_2 fluxes were found (Figure 3.21) in conditions similar to ours of 1) warm temperature, 2) presence of N from fertilization and/or clippings and 3) water availability. A frequently mowed lawn emitted 5.45 t C ha^{-1}, while a lawn infrequently mowed emitted a total of 0.41 t C ha^{-1} in the growing season (Allaire, et al., 2008). Significant temporal variations were observed in CO_2 fluxes ranging from -1.29 to +14.32 µmol CO_2 m^{-2} s^{-1}, with an average of +1.29 µmol CO_2 m^{-2} s^{-1} (Allaire, et al., 2008). The highest CO_2 flux values were observed in high frequently managed turfs (mowing and fertilization) during the warmest period, while low mowing frequency and unfertilized turfgrass had lower NEE. Our data(Figure 3.19 and Table 3.9), show that during the growing season HI had lower NEE compared to lower intensities categories (MI and LI). While Allaire, et al., (2008) measured CO_2 fluxes more than three times higher compared to our case study (Table 3.11).

Also Livesley and colleagues (2010), measuring CO_2 fluxes from August to November (118 days), above urban lawns treated with different water and nutrient management, found fluxes ranging from 4.60 to 18.70 µmol CO_2 m^{-2} s^{-1}, with cumulative NEE of +30.66 ÷ +39.42 t C ha^{-1} yr^{-1}.

Similar NEE values were measured in an American study (Singh, 2007), that estimated CO_2 fluxes in plots of *Festuca arundinacea* and *Lolium perenne*, with and without fungal endophytes colonization, at the Ohio Agricultural and Research development Centre in Wooster, Ohio (United States). Soil surface CO_2 flux, measured between June and September (82 days), resulted higher in the beginning of summer as compared to the end of summer and ranged from 3.15 to 15.15 µmol CO_2 m^{-2} s^{-1}. On the contrary Pataki (2006), founds the high C sequestration rates on her study (NEE of -7.18±0.47 µmol CO_2 m^{-2} s^{-1}). Also a Chilean study (Acuna, et al., 2012) found positive CO_2 assimilation rates on several cool-season grasses during growing season. For some cultivar (*Festuca arundinacea* cv. Cochise and Bingo) the same trend was founded during all the year (15 µmol CO_2 m^{-2} s^{-1}), while in warm-season grasses (e.g. *Cynodon dactylon*) NEE resulted more high in summer (75 µmol CO_2 m^{-2} s^{-1}) and stable during winter dormancy, as confirmed by Dugas and colleagues (1999).

High intensity management surfaces (tees, collar and green) are subjected to a series of different input and disturbances that could differently affect biogeochemical cycles. From one side operations such as coring and mowing are expected to have a negative effect on C accumulation. On the other side, maintenance activities such as irrigation and fertilization, which are strongly boosted in golf courses and particularly in HI surfaces, could enhance C sequestration by accelerating its allocation belowground.

In a newly established managed temperate grassland in Swiss, Amman et al. (2007) found that the average NEE measured by EC over 3 years was affected by management intensities, with intensively managed grassland (4 cuts per year, organic and inorganic fertilization for 200 kg N ha^{-1} yr^{-1}) accumulating 1.47 (±1.30) t of C ha^{-1} yr^{-1} and extensively managed grassland (3 cuts yr^{-1}, no fertilization) emitting 0.57 (+1.30/-1.10) t of C ha^{-1} yr^{-1}. The difference was attributed by the authors to a faster decomposition of SOC under the extensive management stimulated by a deficit of available nutrients in the unfertilised soils. Similarly, Schmitt and colleagues (2009) in alpine grassland found that land use and management affected NEE and its flux components which decreased with decreasing intensity of management and thus site fertility.

The relation between NEE and management categories is also related to the leaf area index (LAI) of the different turfgrasses. High intensity management category included vegetated surfaces with low mowing height , usually between 3-5 mm for green and 8-20 mm for tees and collar (Croce, et al.,2006) while the height of cut ranged from 10 to 40 for MI and was above 80 for LI. These differences in LAI were expected to reduce the photosynthetic activity and consequently the CO_2 uptake of HI surfaces compared to MI and LI.

A possible explanation of the behaviour of the differently managed surfaces is linked to errors related to the adopted methodology. Despite the short duration of the measurement, when the conditions are optimal for photosynthesis (high PAR, warm temperatures and not limiting moisture) the inhibition of photosynthesis caused by the decrease of CO_2 concentration (See section 3.4.1) is proportional to the LAI of the turf. Hence the photosynthesis is underestimated in every categories but, to the very highest degree, in surface with lower mowing intensity, such LI. At the same time when PAR is null and respiration is the only flux, the increase in temperature inside the chamber has similarly a bigger effect enhancing CO_2 fluxes where the LAI is higher. Therefore the sum of two errors causes an underestimation of the NEE greater in low intensities surfaces than in HI.

The study of the relationships between CO_2 fluxes and PAR (Figure 3.16) and temperature (Figure 3.17) do not allow us to solve the issue related to the method limitations because they are based on data already biased by these methodological errors. The CO_2 uptake is mainly controlled by PAR (Figure 3.16), as expected when moisture and temperature are not limiting plant activity (Schulze, et al., 2005), while ecosystem respiration during the night is controlled mainly by temperature (Figure 3.17) always when soil water content is not limiting either because too low or too high. In the present study exponential models well described the relationship with nocturnal CO_2 fluxes on year basis, grouped by management intensities. The increasing temperature sensitivity of the respiration (Q10) associated to increasing management intensities, reflects the different availability of C substrate and nutrients for microbial and plant activity (Davidson, et al., 2006).

As shown in Figure 3.20, the annual NEE resulted in a negligible C source for every maintenance category. Considering turf biomass in constant equilibrium, positive NEE seems to be a direct consequence of the high soil respiration (R_s) rates. In fact, as shown by measurements using the conservation of mass approach (Kaye, et al., 2005), R_s is an important component of the C balance of turfgrass (more than three quarters of the total below ground C allocation). Other studies (Bremer and Ham, 2005) that investigated more components of the C balance of turfgrass confirmed that, even if P_{net} is positive NEP results negative. Excepted for few studies showing positive NEP (Pataki, 2006; Lewis, 2010; Acuna, et al., 2012), all the other studies using SC systems reported negative results, with average and maximum values lower than ours, though with the same order of magnitude.

Our results are comparable with small chambers studies available in the literature, however they are in contrast with those obtained with different measurement approaches (SOC chrono-sequences, ecosystemic-, LCA-, isotopic- and remote-sensing -models) that generally report negative values of NEE, considering turfgrasses as a C sinks (e.g. Dugas, et al., 1999; Qian and Follett, 2002; Milesi, et al., 2005; Singh, 2007; Bartlett and James, 2011). However some studies suggest

that in golf courses soil C pools reach an equilibrium between 30 and 50 years since conversion (Qian and Follett, 2002; Kaye, et al., 2005). Being in the studied turfgrass all the C stocks almost at equilibrium, and considering the likely underestimated C uptake yielded by our approach, we can consider the system at a steady state for C fluxes.

3.5 Conclusion

The present study partially fills the gap of annual NEE estimates of turfgrasses. Several studies adopted the SC method to estimate turfgrass NEE only for short-period surveys. Through the SC approach we show for the first time the seasonal trends of NEE as affected by intensity of management. The positive NEE values showed in Figure 3.19 and 3.20, although small in absolute value, may have been influenced by the alteration of environmental factors at canopy level inside the chamber. We suppose that the increase of temperature and the variation of CO_2 concentration inhibited photosynthesis and increased soil respiration. The agreement of our findings with the available estimates of NEE measured with the same approach seems to confirm the limits of the methodology and suggests that different approaches used for estimating C sequestration may be the main variable for turfgrass assessment as C sink or source.

The intensity of turf management seems to have an influence on its C balance. Carbon dioxide emissions from high intensity surfaces were lower compared to low intensity ones, however this could be the result of methodological errors and need further research to understand which maintenance variables are determinant on turfgrass C sequestration. Despite an exact quantification of the error affecting our estimates is not possible, the study shows a turfgrass system close to equilibrium for C fluxes in the considered period.

3.6 Bibliography

Aber J.D. and Melillo J.M. 2001. Terrestrial ecosystems. Academic Press San Diego, pp. 556.

Acuna A., Villalobos L. and Pastenes C. 2012. Carbon sequestration in selected turfgrass species grown in central Chile: preliminary results. 3rd European Turfgrass Conference, BioforskFokus, pp. 33-34.

Allaire S.E., Dufour-L'Arrivée C., Lafond J.A., Lalancette R. and Brodeur J. 2008. Carbon dioxide emissions by urban turfgrass areas. Canadian Journal of Soil Science 88: 529-532.

Ammann C., Flechard C.R., Leifeld J., Neftel A. and Fuhrer J. 2007. The carbon budget of newly established temperate grassland depends on management intensity. Agriculture, ecosystems & environment 121: 5-20.

Baldocchi D.D., Hincks B.B. and Meyers T.P. 1988. Measuring biosphere-atmosphere exchanges of biologically related gases with micrometeorological methods. Ecology: 1331-1340.

Bandaranayake W., Qian Y.L., Parton W.J., Ojima D.S. and Follett R.F. 2003. Estimation of soil organic carbon changes in turfgrass systems using the CENTURY model. Agronomy Journal 95: 558-563.

Bartlett M.D. and James I.T. 2011. A model of greenhouse gas emissions from the management of turf on two golf courses. Science of the total environment 409: 1357-1367.

Beard J.B. and Green R.L. 1994. The role of turfgrasses in environmental protection and their benefits to humans. Journal of Environmental Quality 23: 452-460.

Bremer D.J. and Ham J.M. 2005. Measurement and partitioning of in situ carbon dioxide fluxes in turfgrasses using a pressurized chamber. Agronomy Journal 97: 627-632.

Chapin III F., Woodwell G., Randerson J.T., Rastetter E.B., Lovett G., Baldocchi D.D., Clark D., Harmon M.E., Schimel D.S. and Valentini R. 2006. Reconciling carbon-cycle concepts, terminology, and methods. Ecosystems 9: 1041-1050.

Chapin III F.S. and Matson P.P.A. 2011. Principles of terrestrial ecosystem ecology, Springer, pp. 529.

Colding J. and Folke C. 2009. The role of golf courses in biodiversity conservation and ecosystem management. Ecosystems 12: 191-206.

Croce P., De Luca A., Falcinelli M., Modestini F. S. and Veronesi F. 2006. Tappeti erbosi, Edagricole, pp. 340.

Curiel Yuste J., Baldocchi D. D., Gershenson A., Goldstein A., Misson L. and Wong S. 2007. Microbial soil respiration and its dependency on carbon inputs, soil temperature and moisture. Global Change Biology 13: 2018-2035.

Davidson E.A., Janssens I.A. and Luo Y. 2006. On the variability of respiration in terrestrial ecosystems: moving beyond Q10. Global Change Biology 12: 154-164.

Dugas W.A., Heuer M.L. and Mayeux H.S. 1999. Carbon dioxide fluxes over bermudagrass, native prairie, and sorghum. Agricultural and Forest Meteorology 93: 121-139.

Federazione Italiana Golf, 2013. Statistiche 2012. URL consulted on 2013 July 11.http://www.federgolf.it/ContentPage.aspx?page=statistiche

Fang C. and Moncrieff J.B. 1998. An open-top chamber for measuring soil respiration and the influence of pressure difference on CO_2 efflux measurement. Functional Ecology 12: 319-325.

Fang C. and Moncrieff J.B. 2001. The dependence of soil CO_2 efflux on temperature. Soil Biology and Biochemistry 33: 155-165.

Flanagan L.B., Wever L.A. and Carlson P.J. 2002. Seasonal and interannual variation in carbon dioxide exchange and carbon balance in northern temperate grassland. Global Change Biology 8: 599-615.

Fortin M.C., Rochette P. and Pattey E. 1996. Soil carbon dioxide fluxes from conventional and no-tillage small-grain cropping systems. Soil Science Society of America Journal 60: 1541-1547.

Hammond R.A. and Hudson M.D. 2007. Environmental management of UK golf courses for biodiversity—attitudes and actions. Landscape and urban planning 83: 127-136.

Hull R.J. 1987. Kentucky bluegrass photosynthate partitioning following scheduled mowing. Journal of the American Society for Horticultural Science, 112:829-834.

Hull R.J. 2000. Mowing, its impact on turfgrass. Turfgrass Trend, pp. 1-8.

Hunt S. 2003. Measurements of photosynthesis and respiration in plants. Physiologia Plantarum 117: 314-325.

Huyler A., Chappelka A.H., Prior S.A. and Somers G.L. 2013. Drivers of soil carbon in residential 'pure lawns' in Auburn, Alabama. Urban Ecosystems: 1-15.

Ilmeteo.it, 2014. Dati meteo della stazione di Verona-Villafranca. URL consulted on 2014 February 21.

http://www.ilmeteo.it/portale/medie-climatiche/Sommacampagna

IPCC. 2013. Summary for Policymakers. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S. K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Kaye J., McCulley R. and Burke I. 2005. Carbon fluxes, nitrogen cycling, and soil microbial communities in adjacent urban, native and agricultural ecosystems. Global Change Biology 11: 575-587.

Kirschbaum M., Eamus D., Gifford R., Roxburgh S. and Sands P. 2001. Definitions of some ecological terms commonly used in carbon accounting. Cooperative Research Centre for Carbon Accounting, Canberra: 2-5.

KMPG. 2011. Golf participation in Europe 2011. KMPG´s Golf Advisory Practice in Europe, Community Middle East and Africa & Golf Business Community, Budapest, Hungary, pp. 6.

Kutzbach L., Schneider J., Sachs T., Giebels M., Nykänen H., Shurpali N.J., Martikainen P.J., Alm J. and Wilmking M. 2007. CO_2 flux determination by closed-chamber methods can be seriously biased by inappropriate application of linear regression. Biogeosciences 4: 1005-1025.

Laghai H.A. and Bahmanpour H. 2012. GIS Application in Urban Green space Per Capita Evaluation (Case study: City of Tehran). Annals of Biological Research 3: 8.

Lal R. 2004. Soil carbon sequestration to mitigate climate change. Geoderma 123: 1-22.

Lewis J.D. 2010. Carbon, nitrogen and water fluxes from turfgrass ecosystems. Department of Horticulture, Forestry, and Recreation Resources-College of Agriculture. Kansas State University, Manhattan, Kansas, pp. 155.

Livesley S.J., Dougherty B.J., Smith A.J., Navaud D., Wylie L.J. and Arndt S.K. 2010. Soil-atmosphere exchange of carbon dioxide, methane and nitrous oxide in urban garden systems: impact of irrigation, fertiliser and mulch. Urban Ecosystems 13: 273-293.

Livingston G.P. and Hutchinson G.L. 1995. Enclosure-based measurement of trace gas exchange: applications and sources of error. Biogenic trace gases: measuring emissions from soil and water: 14-51.

Livingston G.P., Hutchinson G.L. and Spartalian K. 2006. Trace Gas Emission in Chambers. Soil Science Society of America Journal 70: 1459-1469.

Lovett G.M., Cole J.J. and Pace M.L. 2006. Is net ecosystem production equal to ecosystem carbon accumulation? Ecosystems 9: 152-155.

McPherson E.G., Nowak D., Heisler G., Grimmond S., Souch C., Grant R. and Rowntree R. 1997. Quantifying urban forest structure, function, and value: the Chicago Urban Forest Climate Project. Urban ecosystems 1: 49-61.

Milesi C., Running S.W., Elvidge C.D., Dietz J.B., Tuttle B.T. and Nemani R.R. 2005. Mapping and modeling the biogeochemical cycling of turf grasses in the United States. Environmental Management 36: 426-438.

Minelli A., Carnevali M. and Lobietti R. 2008. Golf and landscape: suggestions for special planning in urban areas, pp. 502-508.

Murphy J.T. 2007. Patterns of carbon dioxide and water vapor flux following harvest of tallgrass prairie at different times throughout the growing season. Department of Agronomy-College of Agriculture. Kansas State University, Manhattan, Kansas, pp. 109.

Müller J., Eschenröder A. and Diepenbrock W. 2009. Through-flow chamber CO_2canopy gas exchange system. Construction, microclimate, errors, and measurements in a barley (*Hordeum vulgare* L.) field. Agricultural and Forest Meteorology 149: 214-229.

Nowak D.J., Crane D.E., Stevens J.C., Hoehn R.E., Walton J.T. and Bond J. 2008. A ground-based method of assessing urban forest structure and ecosystem services. Arboriculture and Urban Forestry 34: 347-358.

Nowak D.J., Stevens J.C., Sisinni S.M. and Luley C.J. 2002. Effects of urban tree management and species selection on atmospheric carbon dioxide. Journal of Arboriculture 28: 113-122.

Pataki D.E. 2006. Controls on C and N cycling in a southern California urban turfgrass ecosystem. Soil Carbon and California's Terrestrial Ecosystems 2005221.

Pickering N.B., Jones J.W. and Boote K.J. 1993. Evaluation of the portable chamber technique for measuring canopy gas exchange by crops. Agricultural and forest meteorology 63: 239-254.

Qian Y. and Follett R.F. 2002. Assessing soil carbon sequestration in turfgrass systems using long-term soil testing data. Agronomy Journal 94: 930-935.

Raich J.W. and Schlesinger W.H. 1992. The global carbon dioxide flux in soil respiration and its relationship to vegetation and climate. Tellus B 44: 81-99.

Razmjoo K., Imada T. and Kaneko S. 1996. A comparative study of the effects of mowing height on Kentucky bluegrass (*Poa pratensis* L.) cultivars. Journal of Turfgrass Management 1: 33-42.

Russo A., Escobedo F.J., Timilsina N., Schmitt A.O., Varela S. and Zerbe S. 2014. Assessing urban tree carbon storage and sequestration in Bolzano, Italy. International Journal of Biodiversity Science, Ecosystem Services & Management: 1-17.

Saito O. 2010. Measuring the lifecycle carbon footprint of a golf course and greening the golf industry in Japan. 4th International Conference on Sustainability Engineering and Science.

Schmitt M., Bahn M., Wohlfahrt G., Tappeiner U. and Cernusca A. 2009. Land use affects the net ecosystem CO2 exchange and its components in mountain grasslands. Biogeosciences Discussions 6: 13.

Schulze E.D., Beck E. and Müller-Hohenstein K. 2005. Plant ecology. Springer, pp. 702.

Selhorst A. and Lal R. 2013. Net Carbon Sequestration Potential and Emissions in Home Lawn Turfgrasses of the United States. Environmental management 51: 198-208.

Singh M.H. 2007. Soil Organic Carbon pools in turfgrass systems in Ohio. The Ohio State University, pp. 160.

Sorace A. and Visentin M. 2007. Avian diversity on golf courses and surrounding landscapes in Italy. Landscape and urban planning 81: 81-90.

Steduto P., Çetinkökü Ö., Albrizio R. and Kanber R. 2002. Automated closed-system canopy-chamber for continuous field-crop monitoring of CO_2 and H_2O fluxes. Agricultural and forest meteorology 111: 171-186.

Tanner R.A. and Gange A.C. 2005. Effects of golf courses on local biodiversity. Landscape and Urban planning 71: 137-146.

The Cambridge Dictionary Online 2013. Golf course in «The Cambridge Dictionary Online», URL consulted on 2013 August 20.http://dictionary.cambridge.org/dictionary/british/golf-course?q=golf+course

Turgeon A. 1985. Turfgrass management. Reston Publishing Company, pp. 396.

Venturelli F. and Virli L. 1995. Invito alla botanica: con esercitazioni di laboratorio. Zanichelli, pp. 528.

Wagai R., Brye K. R., Gower S. T., Norman J. M. and Bundy L. G. 1998. Land use and environmental factors influencing soil surface CO_2 flux and microbial biomass in natural and managed ecosystems in southern Wisconsin. Soil Biology and Biochemistry 30: 1501-1509.

Wagner S.W. and Reicosky D.C. 1992. Closed-chamber effects on leaf temperature, canopy photosynthesis, and evapotranspiration. Agronomy journal 84: 731-738.

Walter H. 1960. Klimadiagramm-weltatlas. Veb Gustav Fischer Verlang Jena.

Weinstock L., Kender W. and Musselman R. 1982. Microclimate within open-top air pollution chambers and its relation to grapevine physiology. Journal American Society for Horticultural Science 107.

Woodwell G.M. and Whittaker R.H. 1968. Primary production in terrestrial ecosystems. American Zoologist 8: 19-30.

Zhou X., Wang X., Tong L., Zhang H., Lu F., Zheng F., Hou P., Song W. and Ouyang Z. 2012. Soil warming effect on net ecosystem exchange of carbon dioxide during the transition from winter carbon source to spring carbon sink in a temperate urban lawn. Journal of Environmental Sciences 24: 2104-2112.

CHAPTER 4
Conclusion

The present study reached the purpose of clarifying the factors affecting the C cycle in turfgrasses systems and the underlying mechanisms. The exhaustive analysis of the literature data coupled with the direct estimation of the NEE in a golf course over one year improve our understanding on the C sequestration potential of such systems.

Through the literature analysis we identified the factors affecting C sequestration on turfgrass. Natural drivers, such as environmental conditions, site peculiarity and former land use affect turfgrass C sequestration together with other anthropogenic drivers, such as fertilization, irrigation and mowing, with the latter that may often prevails. Because of the high variability of the above mentioned factors no univocal guidelines for the best agronomical practices for a sustainable management can be given. However, the present study improved the understanding of the interaction among environmental and human factors, and provided some operative indications for balancing interventions in order to respect the environment, satisfy human needs and limit the costs.

Chapter 2 showed that turfgrasses are important C pools, but their ability to act as C-sinks depends on several factors, such as previous land use, soil texture, turfgrass typology, species composition, seasonal and climatic trends, characteristics and intensity of management.

We highlighted how the methodological approach adopted to study C fluxes in the 23 empirical studies so far published on the subject, have influenced the attribution of a system to the C-sink or -source categories. Generally, the studies investigating SOC dynamics over a wide temporal scale showed that turfgrass acts as a C-sink while studies estimating the NEE by small-chamber approach showed urban lawns as C sources, even during the growing season. Such apparent contradiction is likely the results of some methodological limits of the small chamber enclosure approach compared to other studies, that have been widely discussed in Chapter 3.

SOC models and chrono-sequences, even if generally derived by datasetsoriginating from direct measurements, only partially help to understand turfgrass behaviour, as they fail to capture the plant-soil mechanism and interactions.

In Chapter 3 the small chamber enclosure approach that we adopted lead to an estimation of the NEE as affected by intensity of management. The study showed a turfgrass system close to the equilibrium for C fluxes.

This is the first annual estimation of C balance showing how different intensities of management can influencethe seasonal trends of NEE.

The methodological issues that may have affected the measurementshave been exhaustively discussed and likely lead to an underestimation of the C uptake of the system.

The agreement of our findings with the available estimates of NEE measured with the same approach seems to confirm the limits of the methodology.

The estimate of hidden anthropogenic C costs, related to the turfgrass maintenance, completed the annual C balance of the golf course hole considered. Even if maintenance activities increase C sequestration, they are at the same time responsible for C emissions due to fossil fuel combustion. Turfgrass emissions due to maintenance operations resulted 0.846, 0.134 and 0.146 t CE ha^{-1} yr^{-1} respectively for HI, MI and LI. However, adequate choices at turfgrass establishment, maintenance operations planning and machinery turn-over can decrease the amount of GHGs emitted.

In conclusion, turfgrass may represent a C source for golf courses and more generally for urban and sub-urban landscapes. However, turfgrasses are part of a wider system (including shrubs, trees and more wild areas). The sustainability of green areas depends primarily on the design that, regarding the specific functions, should guarantee an adequate ratio between turfgrass and shrubs-trees areas. Further studies need to be conducted to consider the C potential of trees and shrubs in order to complete the C budget of green urban areas and contrast GHGs emissions related to maintenance activities.

APPENDIX

Effect of management on hidden carbon costs in a golf course in the province of Verona

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A.1 Introduction

Terrestrial ecosystems play an important role in global warming mitigation, affecting the concentration of greenhouse gases (GHGs) in atmosphere (IPCC, 2013). Usually natural ecosystem, such as grassland and forests, serve as C sink, while agricultural land and urban ecosystems act as a C source (Del Grosso, et al., 2009). In spite of the increasing number of studies investigating the C sequestration potential of turfgrasses, their role in acting as C sources or sink is not fully understood (See Chapter 2 in this study). Several studies reported that turfgrass C sequestration potential is positively conditioned by intensive management practices (See also Chapter 3 in this study) but on the other hand, obviously, management practices increase the energy costs of the system. Falk (1976) estimated that the overall energy input for management of home lawn was 578 Kcal m^{-2} yr^{-1}. Despite the efficiency of machinery and production processes have been enhanced in last 40 years, we suppose that the maintenance intensity of a golf course turfgrass require nowadays higher levels of energy input, due to the increasing standards of playability.

These input, while providing the aesthetics and utilitarian functions of turfgrasses, are largely based on fossil fuel consumption (e.g. mowing and cultural practices, irrigation, fertilizers and agro-chemicals production, transportation, storage and application) and can affect GHGs emissions from the soil (e.g. increased CH_4 and N_2O soil emissions after fertilization). These inputs therefore have hidden C costs that should be accounted to estimate the C balance of a managed system.

Turfgrass global warming potential (GWP), has been studied in relation to the principal GHGs (CO_2, CH_4 and N_2O) emissions, accounted as CO_2 equivalents. As an example, application of N with fertilizers leads to important N_2O emissions, as confirmed by several studies (Bremer, 2006, Groffman, et al., 2009, Livesley, et al., 2010, Zhang, et al., 2013). The recorded emissions from frequently fertilized turfgrasses range from 0.5 to 6.4 kg N ha^{-1} yr^{-1} (Zhang, 2012), with values similar to agricultural land (Townsend-Small and Czimczik, 2010) or even greater (Kaye, et al., 2004).

According to Staples (2009), the largest contributors to the C footprint of turfgrasses are fossil fuels and electricity to power the landscaping machinery and the irrigation system, fertilizers to increase and maintain turf health, and the embedded C footprint in the landscaping machinery.
Three studies estimated GWP balances related to golf courses turfgrasses. Selhorst and Lal (2011) outlined the estimates of C emission from data collected from 11 golf courses in central Ohio (United States). Bartlett and James (Bartlett and James, 2011) adopted a modelling approach based upon a modified life cycle analysis (LCA), calculating emissions from a wide dataset adapted to two golf courses in the UK. Whereas a study on Northern California (United States) (Valenti, 2010) created a golf course inventory of resources used for operations and maintenance, analyzing it with an economic input-output Life Cycle Analysis (EIO-LCA) model. Other studies estimated GHGs emissions in different turfgrass typologies such as home lawns (Selhorst and Lal, 2013), ornamental lawns and athletic fields (Townsend-Small and Czimczik, 2010).
Our aim is 1) to estimate CO_2, CH_4 and N_2O direct emissions due to fossil fuel consumption for turfgrass upkeep, and 2) to outline a C balance, comparing annual GHGs emissions with C sequestration rate shown in Chapter 3.

A.2 Materials and methods

For the GHGs inventory, maintenance activities carried out at the 18th hole of the Golf Club Verona from September 1st 2012 to August 31st 2013 were considered. The superintendent monitored the cultural operations effectuated in the different playing areas, surveying the annual amount of engine working times. The fuel consumption for each playing areas of the 18th hole has been calculated by the timing employed by each singular cultural operation considered: mowing and brush cutting, fertilization and agro-chemicals application, verti-cutting, topdressing and over-seeding. Emissions due to energy consumption by the irrigation system, was not considered because the lack of available data. Other indirect emissions not considered in the calculation are: emissions derived from the embedded C footprint in the landscaping machinery; emissions connected to repairing divots, raking bunkers, leaves blowing and other negligible operations connected to turfgrass upkeep; emissions connected to tree pruning and maintenance of non-playing areas; emissions from soil of CH_4 and N_2O derived from fertilization; emissions related to energy use of service buildings (club house, machineries workshop,..).

A.2.1 Theory of operations

2006 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006) provide the fundamental methodologies for estimating greenhouse gas emissions from off-road vehicles, including agriculture and turf machinery.

Mobile sources produce direct GHG emissions of carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) from the combustion of various fuel types, as well as several other pollutants such as carbon monoxide (CO), non-methane Volatile Organic Compounds (NMVOCs), sulphur dioxide (SO_2), particulate matter (PM) and oxides of nitrate (NO_x), which contribute to local or regional air pollution (IPCC, 2006). However, for this inventory, CO_2, $CH_4$and N_2O emissions were considered. Those can be estimated from either the fuel consumed or the distance travelled by the vehicles. In general, the first approach is more appropriate for CO_2 estimate, while the second is more appropriate for CH_4 and N_2O (IPCC, 2006).

Emissions of CO_2 are calculated based on the amount and type of fuel combusted and its C content. The equation A.1 calculates CO_2 emissions (Kg) by multiplying estimated fuel consumption for a fixed CO_2 emission factor:

$$Emissions = \sum_a \quad (Fuel_a \times EF_a) \qquad \textbf{(eq. A.1)}$$

where:

$Fuel_a$ = fuel (TJ) = NFCVs (TJ/l) x Fuel volume (l)

EF_a = emission factor (kg/TJ).

a = type of fuel

Calculation of $Fuel_a$

In energy statistics and other energy data compilations, production and consumption of fuels are specified in physical units, e.g. in kilograms or liters (IPCC, 2006a). To convert these data to common energy units for the estimate of CO_2 emissions as shown in equation A.1, it requires calorific values, and specifically net calorific values (NCVs). Net calorific fuel values (TJ/l) are calculated as:

$$NCFVs = NCVs \times d \times 10^{-6} \qquad \textbf{(eq. A.2)}$$

where:

NCVs[1] is net calorific values: 44.3 for motor gasoline; 43.0 for diesel oil)

d: density: 0.84 for motor gasoline; 0.753 for diesel oil

[1] Default Net calorific value (NCVs) from Table 1.2 of 2006 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006a).

Consequently NFCVs result 37.212 x 10^{-6}TJ/l for gasoline and 32.379 x 10^{-6}TJ/l for diesel oil.

Calculation of CO_2 emission factor (EF_a)

Combustion processes are optimized to derive the maximum amount of energy per unit of fuel consumed, hence delivering the maximum amount of CO_2. Efficient fuel combustion ensures oxidation of the maximum amount of C available in the fuel. Carbon dioxide emission factors for fuel combustion are therefore relatively insensitive to the combustion process itself and hence are primarily dependent only on the C content of the fuel (IPCC, 2006a).

Only a small part of the fuel C entering the combustion process escapes oxidation (1%) thus default emission factors are derived on the assumption of 100% oxidation. The effective CO_2 emission factor (EF_a) (kg/TJ) is calculated as:

$$\mathrm{EF_a} = \text{default C content} \times \text{default C oxidation factor} \times \mathrm{CO_2}\ \text{produced from C combustion} \times 3.67 \times 10^3 \qquad \textbf{(eq. A.3)}$$

where:

default C content[2] (kg/GJ): 18.9for gasoline; 20.2 for diesel oil.

default C oxidation factor (0.99-1): 1

CO_2 produced from C combustion (44/12, kg/Kg) ≈ 3.67 kg/Kg

Therefore EF_a is 69.300Kg/TJ for gasoline and 74.100 Kg/TJ for diesel oil.

Calculation of CH_4 and N_2O emissions

Concerning CH_4 and N_2O, their emission rates depend largely upon the combustion and emission control technology present in the vehicles. The default emission factors used for CH_4 and N_2O emission estimate are based on European off-road mobile sources and machinery. In general, off-road vehicles, including turf machinery, do not have emission control catalysts installed. Properly operating catalysts convert nitrogen oxides to N_2O and CH_4 to CO_2.

Being the information available on machinery limited, we used default fuel-based emission factors that do not specify vehicle technology, even if this method makes the estimate highly uncertain. The equations A.1, A.2 and A.3 can be used as well for CH_4 and N_2O.

From equations A.1, A.2 and A.3, CO_2 emitted for each litre of fuel consumed is 2.581 and 2.400 kg CO_2eq respectively for gasoline and diesel, CH_4 emitted for each litre of fuel consumed is 0.074 (gasoline) and 0.003 kg CO_2eq respectively for

[2] Default carbon content from Table 1.4 of 2006 IPCC Guidelines for National Greenhouse Gas Inventories, (IPCC, 2006a).

gasoline and diesel[3], N_2O emitted for each litre of fuel consumed is 0.001 and 0.023 kg CO_2eq respectively for gasoline and diesel[4].

A.3 Results and discussion

Annual fuel consumption for turfgrass maintenance is mainly caused by mowing activity (418 l) while other cultural operations less frequent consume significantly less (Table A.1 and A.2). The highest consumption occurred in rough and fairway, due to their relative wide extension, while consumption for green mowing is high despite the scarce surface, due to the frequency of cutting.

Table A.3 reports the emissions of the 3 main GHGs due to mowing, divided by playing surfaces. Methane (CH_4) emissions resulted negligible compared to CO_2 emissions (1013 kg CO_2eq), while N_2O resulted 10% of the latter. For the 18th hole, including 1.8268 ha of turfgrass maintained at golf course standards, the overall GHGs release due to fossil fuel combustion related to mowing was 1121 kg CO_2eq yr^{-1}.

Only green, tees, fairways and bunkers were affected by extra-mowing operations (Table A.2), and among them, fertilizers application was the most frequent. However, of the 25 annual applications, more than half (13) are executed by hand with no energy costs, therefore topdressing resulted the extra-mowing operation that consumed more fuel (21 l) (Table A.2).

In spite of the small relative surface, the high frequency of mowing caused important emissions (303 kg CO_2eq yr^{-1}) in the high intensity maintenance category (green, tees and collar; HI) if compared to the more extended areas of the hole pertinent to the medium and low intensities of maintenance categories (MI and LI) that released respectively 406 and 411 kg CO_2eq yr^{-1} (Table A.3). Observing overall emissions per hectare, HI playing areas had the highest values, while LI exceeded MI emissions (146 to 116 kg CO_2eq yr^{-1}). This can be attributed to two factors: the high consumption required for mount mowing (Toro Sidewinder 3500), due to the roughness of the surface, and the necessity to mowing bunkers edging and to brush-cutting trees surrounding, which required a lot of time and fuel consumption despite they constitute small surfaces. Emissions related to extra-mowing operations affected mainly fairway, due to the high surface (Table A.4).

The overall GHGs balance shows that 6% of the hole (green and tees) emitted 28% of the GHGs (Table A.5). After green, tee and collar, the playing area that releases

[34] Default Emission Factor from Table 3.3.1 of 2006 IPCC Guidelines for National Greenhouse Gas Inventories, (IPCC, 2006)

more GHGs was mount. Although it was mowed less frequent than fairway and semi-rough, it required more energy for mowing on equal surfaces.

In conclusion, we estimate that one hectare of turfgrass maintained with golf course standards emitted 0.185 t CE yr^{-1} (carbon equivalent) due to maintenance activities.

Selhorst and Lal (2011) estimated that total C emission from turfgrass maintenance practices were 0.289 t CE ha^{-1} yr^{-1}, with 75% attributed to mowing and 25% equally divided between agro-chemicals and fertilizers applications. The CRANTURFC model adopted by Bartlett and James (2011), considering two golf courses with different management policies and ratio among surfaces, showed considerably higher values(1.987 and 2.584 t CE ha^{-1} yr^{-1} respectively), however, compared to the previous study, lubricant use and production, plus transportation of agro-chemical and fertilizers were included. A five-year inventory in three Californian golf courses (Valenti, 2010) permitted the modelling of C footprint related to operation and maintenance. The total fossil fuel ranged between 0.170 and 0.350 t CE ha^{-1} yr^{-1}. However, because of the uncertain of the considered practices, it results difficult to compare these data with the previous ones.

Furthermore, emissions related to turfgrass with lower maintenance degrees, such as urban ornamental turfs and home lawns, resulted similar or greater to that one showed in golf courses (respectively 0.254 and 0.329 t CE ha^{-1} yr^{-1}) (Townsend-Small and Czimczik, 2010; Selhorst and Lal, 2013).These data suggest that, beside the variables related to the maintenance degree, the estimation methodology may considerably influence final results.

As for most of the studies, we aim to understand the ratio between the released C and the sequestrated one, outlining a balance that considers natural and anthropogenic fluxes.

Using maintenance categories presented in Chapter 3, turfgrass emissions due to maintenance operations resulted 0.846, 0.134 and 0.146 t CE ha^{-1} yr^{-1} respectively for HI, MI and LI. Also NEE resulted positive, though it can be considered almost in steady state, with C emissions of 0.036, 0.057 and 0.071 t C ha^{-1} yr^{-1} respectively for HI, MI and LI.

Table A.1: Characteristics of mowing activity on 18th hole during the year of monitoring. Machineries are provided with diesel (die.) or gasoline (gas.) engine. (*) Multiple rotary mower connected to tractor Carraro TRF 7400 – 74 hp.

Playing area	Surface (ha)	Surface (percentage)	Machinery	Mowing counts	Annual consumption (l)
green	0.0541	3.0%	John Deere A 2500 – 16 hp (die.)	225	68
tee	0.0512	2.8%	Toro Greenmaster 3100 – 14 hp (gas.)	127	38
collar	0.012	0.7%	Toro Greenmaster 3100 – 14 hp (gas.)	113	8
fairway	0.6675	36.5%	Jacobsen LF 3400 – 34 hp (die.)	106	106
semi rough	0.28	15.3%	Ransomes 350 D – 65 hp (die.)	91	46
mount	0.012	0.7%	Toro Sidewinder 3500 (die.)	74	7
rough	0.75	41.1%	Rotomec flex 2003 A360 (40 cv)* (die.)	42	126
bunker edging	-	-	Toro 3040 Sand (gas.)	4	8
trees surrounding	-	-	Kubota f3560 (die.)	4	12
Total	**1.8268**	**100%**		**786**	**418**

Table A.2: Characteristics of cultural operations on 18th hole, excepted for mowing, during the year of monitoring. Machineries are provided with diesel (die.) or gasoline (gas.) engine. (*) Equipment connected to tractor Carraro TRF 7400 – 74 hp.

Playing area	Machinery	Cultural practices counts	Annual consumption (l)
green		**37**	**17.2**
fertilization	by hand	13	0
topdressing	John Deere Pro gator (die.)	9	9
agroch. application	John Deere Pro gator (die.)	12	6
verticutting	Toro Greenmaster 3100 – 14 hp (gas.)	3	1.5
tee		**15**	**7.5**
fertilization	John Deere Pro gator (die.)	6	3
agroch. application	John Deere Pro gator (die.)	9	4.5
fairway		**10**	**23.5**
fertilization	Fertilizer spreader Vicon* (die.)	6	6
topdressing	Sand spreader Valentini (50 cv)* (die.)	2	12
over-seeding	Over-seeder Vredo Supercompact (65 cv)* (die.)	1	5
verticutting	Corer Vertidrain 7117 (30 cv)* (die.)	1	0.5
bunker		**1**	**0.5**
agroch. application	John Deere Pro gator (die.)	1	0.5
Total		**63**	**48.7**

Table A.3: Emissions distributed per surface and overall related to mowing activity during the year of monitoring. CH_4 global warming potential (GWP) factor is 25, N_2O GWP factor is 298.

MOWING GHGs BALANCE					
Playing area	CO_2 Emissions (kg CO_2eq)	CH_4 Emissions (kg CO_2eq)	N_2O Emissions (kg CO_2eq)	Tot. Emissions (kg CO_2eq)	Tot. Emissions/ha (t CE ha^{-1} yr^{-1})
green	162.0	0.23	18.63	181	0.698
tee	98.3	2.84	0.84	102	
collar	19.4	0.56	0.17	20	
fairway	254.4	0.36	29.25	284	0.116
semi rough	109.2	0.15	12.56	122	
mount	17.8	0.02	2.04	20	0.146
rough	302.4	0.42	34.77	338	
bunker edging	20.6	0.60	0.18	21	
trees surrounding	28.8	0.04	3.31	32	
Total	**1013.32**	**5.22**	**101.75**	**1121**	

Table A.4: Emissions distributed per surface and overall related to cultural operations during the year of monitoring. CH_4 global warming potential (GWP) factor is 25, N_2O GWP factor is 298.

Playing area	CO_2 Emissions (kg CO_2 eq)	CH_4 Emissions (kg CO_2 eq)	N_2O Emissions (kg CO_2 eq)	Total emissions (kg CO_2 eq)
green	**39.9**	**0.162**	**4.17**	**44.21**
fertilization	-	-	-	-
topdressing	21.6	0.030	2.48	24.12
agroch. application	14.4	0.020	1.66	16.08
verticutting	3.9	0.112	0.03	4.02
tee	**18.0**	**0.025**	**2.07**	**20.10**
fertilization	7.2	0.010	0.83	8.04
agroch. application	10.8	0.015	1.24	12.06
fairway	**56.4**	**0.079**	**6.49**	**62.97**
fertilization	14.4	0.020	1.66	16.08
topdressing	28.8	0.040	3.31	32.16
over-seeding	12.0	0.017	1.38	13.40
verticutting	1.2	0.002	0.14	1.34
bunker	**1.2**	**0.002**	**0.14**	**1.34**
agroch. application	1.2	0.002	0.14	1.34
Total	**115.5**	**0.268**	**12.87**	**128.62**

Table A.5: Overall emissions including all the cultural practices executed on 18th hole during the considered period of monitoring. Emissions for playing areas (percentage) and emissions per hectare related to rough include mowing of bunker edging and brush cutting around the trees.

MAINTENANCE ACTIVITIES GHG BALANCE					
Playing area	Surface (percentage)	Total Emissions (kg CO_2eq yr^{-1})	Emissions for playing areas (percentage)	Total Emissions/ha (t CE ha^{-1} yr^{-1})	Tot. Emissions/ha (t CE ha^{-1} yr^{-1})
green	3%	225	18%	1.124	
tee	3%	122	10%	0.645	**0.846**
collar	1%	20	2%	0.454	
fairway	37%	347	28%	0.140	**0.134**
semi rough	15%	122	10%	0.118	
mount	1%	20	2%	0.446	
rough	41%	339			**0.146**
bunker edging	-	21	31%	0.141	
trees surrounding	-	32			
Total	**100%**	**1249**	**100%**	**0.185**	

A.4 Conclusion

The evaluating criterion to judge the playability of a golf course is mainly based on the turfgrass quality that often reflects the maintenance degree. However, different agronomic policies, that include more or less frequent cultural practices, may obtain similar quality results (Minelli, et al., 2014) reducing at the same time GHGs emissions. The identification of the weight that singular emission components have on the balance of GHGs may be a good indication for determining the agronomic strategies to reach C neutrality.

From the results of our hidden C costs analysis, we can extrapolate that the most influent variables on the GHGs emissions in a golf course are: 1) the degree of maintenance of the golf course, deduced by the playability quality; 2) the climate and soil characteristics, that affect the turfgrass growth and then the cultural practices; 3) the design of the course, that determines the ratio among different playing areas, and 4) the obsolescence of the machinery fleet, that influences the combustion efficiency.

Even if anthropogenic practices have shown to increase C sequestration on turfgrass ecosystems (See Chapter 2), the C released from turfgrass exceed that sequestered. However, we wish to underline that the C footprint of turfgrasses is not representative of the whole golf course, as trees and shrubs are important components for mitigating C emissions and reaching C neutrality.

A.5 Bibliography

Bartlett M.D. and James I.T. 2011. A model of greenhouse gas emissions from the management of turf on two golf courses. Science of the total environment 409: 1357-1367.

Bremer D.J. 2006. Nitrous Oxide Fluxes in Turfgrass. Journal of environmental quality 35: 1678-1685.

Del Grosso S.J., Ojima D.S., Parton W.J., Stehfest E., Heistemann M., DeAngelo B. and Rose S. 2009. Global scale DAYCENT model analysis of greenhouse gas emissions and mitigation strategies for cropped soils. Global and Planetary Change 67: 44-50.

Falk J.H. 1976. Energetics of a suburban lawn ecosystem. Ecology 57: 141-150.

Groffman P.M., Williams C.O., Pouyat R.V., Band L.E. and Yesilonis I.D. 2009. Nitrate leaching and nitrous oxide flux in urban forests and grasslands. Journal of environmental quality 38: 1848-1860.

IPCC. 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Institute for Global Environmental Strategies, Hayama, Kanagawa, Japan, Vol. 2, Energy, N.o Chapter 3, Mobile sources, pp. 78.

IPCC. 2006a. 2006 IPCC Guidelines for National Greenhouse Inventories. Institute for Global Environmental Strategies, Hayama, Kanagawa, Japan , Vol. 2, Energy, N.o Chapter 1, Introduction, pp. 29.

IPCC. 2013. Summary for Policymakers. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S. K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Kaye J.P., Burke I.C., Mosier A.R. and J P.G. 2004. Methane and nitrous oxide fluxes from urban soils to the atmosphere. Ecological Applications 14: 975-981.

Livesley S.J., Dougherty B.J., Smith A.J., Navaud D., Wylie L.J. and Arndt S.K. 2010. Soil-atmosphere exchange of carbon dioxide, methane and nitrous oxide in urban garden systems: impact of irrigation, fertiliser and mulch. Urban Ecosystems 13: 273-293.

Minelli A., De Luca A., Croce P., Cevenini L. and Zuffa D. 2014. Transition from cool-season to warm-season grass: environmental effects in a golf course in the North of Italy. European Journa of Turfgrass in press.

Selhorst A. and Lal R. 2013. Net Carbon Sequestration Potential and Emissions in Home Lawn Turfgrasses of the United States. Environmental management 51: 198-208.

Selhorst A.L. and Lal R. 2011. Carbon budgeting in golf course soils of Central Ohio. Urban Ecosystems 14: 771-781.

Staples A. 2009. Golf Course Energy Use, Part 1: Energy Generation and Delivery. Golf Course Management: 96-102.

Townsend-Small A. and Czimczik C.I. 2010. Carbon sequestration and greenhouse gas emissions in urban turf. Geophysical Research Letters 37, pp.5.

Valenti A. 2010. Carbon Footprint Analysis of Operation and Maintenance of Northern California Golf Courses, pp. 18.
http://nature.berkeley.edu/classes/es196/projects/2010final/ValentiA_2010.pdf

Zhang Y. 2012. Simulated carbon and nitrogen dynamics in turfgrass systems using the DAYCENT model-Chapter2: Simulation of N_2O emissions and estimation of global warming potential in turfgrasses using the DAYCENT model. Department of Horticulture and Landscape Architecture. Colorado State University, Fort Collins, Colorado, pp. 78.

Zhang Y., Qian Y., Bremer D.J. and Kaye J.P. 2013. Simulation of Nitrous Oxide Emissions and Estimation of Global Warming Potential in Turfgrass Systems Using the DAYCENT Model. Journal of Environmental Quality 42:1100-1108.